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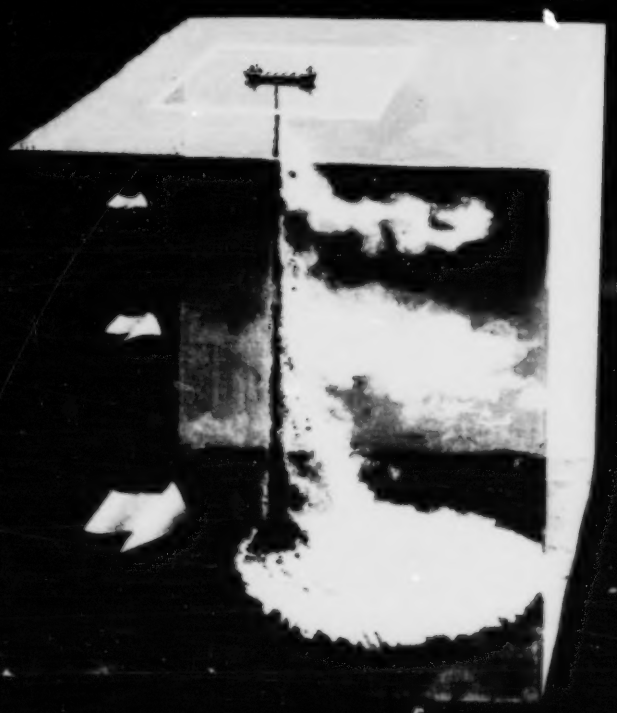

7th INTERNATIONAL OCEAN DISPOSAL SYMPOSIUM

21-25 SEPTEMBER, 1987
WOLFVILLE, NOVA SCOTIA
CANADA

7ième SYMPOSIUM INTERNATIONAL SUR LE REJET DE DÉCHETS EN MER

21 AU 25 SEPTEMBRE 1987
WOLFVILLE (NOUVELLE-ÉCOSSE)
CANADA

Environment Canada Environnement Canada

Canada

PROCEEDINGS

COMPTES RENDUS

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Canada

Du 21 au 25, septembre 1987
Wolfville, Nouvelle-Écosse
Canada

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PREFACE

As co-chairmen, we were pleased to welcome the 7th International Ocean Disposal Symposium to Canada and Nova Scotia. There were over 100 participants representing 19 countries in attendance.

The purpose of the symposia is to bring together, about every two years, scientists, engineers, and program managers to exchange ideas and present the results of studies related to the general theme of disposal of wastes in the marine environment. To that end, the executive committee organized seven sessions to accommodate the 56 verbal presentations and four poster presentations. In addition, interested agencies or other ogranizations were given the opportunity to make presentations on their activities related to the conference theme.

Sessions covered the following topics:

Industrial Waste and Sewage Sludge in the Oceans
Ocean Outfalls
Dredged Material Disposal
Pathways and Fates of Wastes in Marine Environments
Biological Effects
Incineration at Sea
Risk Assessment in Marine Waste Management

Proceedings of past symposia have been subject to careful review and critique, but this has resulted in their being criticized for slow turn-around time. Therefore we elected to provide the manuscripts in the enclosed format. These proceedings are "camera-ready" versions as supplied by the authors. The manuscripts were proof-read, but were not altered editorially nor were they reviewed by external referees. We have encouraged all authors to submit the papers to refereed journals. In addition, we are making arrangements with Marine Pollution Bulletin for another special supplement to be made up of select papers from this conference.

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PRÉFACE

Nous avons été très heureux, en tant que coprésidents, d'accueillir au Canada et en Nouvelle-Écosse le 7e Colloque international sur l'immersion en mer. Plus de 100 délégués de 19 pays y ont participé.

L'objet du colloque, qui se répète à tous les deux ans environ, est de réunir les scientifiques, les ingénieurs ainsi que les gestionnaires de programmes pour l'échange de points de vue et la communication de résultats d'études sur le thème général de l'élimination des déchets dans le milieu marin. À cette fin, le comité exécutif a organisé, outre quatre séances d'affichage, sept séances pour la présentation des 56 présentations orales. De plus, les organismes intéressés ont eu l'occasion de présenter leurs activités qui se rapportaient au thème du colloque.

Les séances ont porté sur les sujets suivants:

Boues résiduaires et déchets industriels dans les océans
Émissaires marins
Élimination des déblais de dragage
Cheminement et devenir des déchets en milieu marin
Effets biologiques
Incinération en mer
Évaluation des risques liés à la gestion des déchets en milieu marin

Les comptes rendus des colloques antérieurs faisaient l'objet d'une révision critique minutieuse, mais la lenteur de la parution avait suscité des observations négatives. C'est pourquoi nous avons choisi de publier les manuscrits selon la présentation actuelle. Il s'agit des versions prêtes à photocopier fournies par les auteurs. Les épreuves ont été lues mais n'ont pas été modifiées en essence ni révisées par des lecteurs de l'extérieur. Nous avons incité tous les auteurs à soumettre leurs exposés à des périodiques qui disposent de comités de lecture. En outre, nous sommes en pourparlers avec le Marine Pollution Bulletin pour que soit publié un autre supplément spécial constitué de présentations orales choisies parmi celles du colloque.

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TABLE OF CONTENTS

Page

INTRODUCTORY PAPER
Role of Ocean Disposal in Canada's Waste Management Strategy. 1
J. Karau

SESSION I: INDUSTRIAL WASTE AND SEWAGE SLUDGE IN THE OCEANS

Stabilized Incineration Residue: A Potential Material for Artificial Reef Construction. 6
F.J.Roethel, V.T. Breslin, V.P. Schaeperkoetter and P. M. J. Woodhead.

Short-Term Effects of Stabilized Oil Ash Reefs on the Marine Benthos. 28
W.G. Nelson, P.M. Navratil, D.M. Savercool, and F.E. Vose.

Polycyclic Aromatic Hydrocarbons Associated with an Artificial Reef Constructed of Coal and Ash Waste Products. 47
R.A. Frease and J.G. Windsor Jr.

Issues and Concerns Surrounding the Ocean Disposal of Fish Processing Wastes and a Case Study. 57
A. McIver and D. Aggett.

Phaseout and Closure of the 12-Mile Sewage Sludge Dump Site: A Case Study on Ecosystem Response. 68
A.L. Studholme.

Selection of an Ocean Dumpsite in the Canadian Beaufort Sea. 93
G.A. Packman.

Results of the Seabed Working Group Assessment of Subseabed Disposal of High-Level Radioactive Wastes. 105
K.R. Hinga and D.R. Anderson.

Alberni Inlet, British Columbia: A Multi-Use Fjord System with Fisheries Resources and a Large Waste-Disposal Component. 107
M. Waldichuk.

Environmental Impact of Solid Waste Disposal on the Coastal and Marine Ecosystems of Kenya. 137
D. N. Kinyanjui.

Page

A New Technique for the Rapid Identification of Sewage and Sludge in the Ocean Using Light Microscopy 139
R. Freudenthal.

SESSION II:OCEAN OUTFALLS

The Environmental Impact of Two Deep Sea Effluent Disposal Systems in Natal, South Africa. 151
T. P. McClurg.

Fish Community Structure on a Deep Ocean Outfall: Barber's Point, Oahu, Hawaii. 161
A.R. Russo.

A Comparison of the Effects of Deep-Ocean Sewer Outfalls on Benthic Infaunal Communities of Oahu, Hawaii. 162
W.G. Nelson, J.H. Bailey-Brock, E.A. Kay, D.A. Davis, M. E. Dutch and R.K. Kawamoto.

A Study of Beach Exposure from an Offshore Marine Outfall. 201
S.R. Rives and J. J. Gift.

Effects of Sewage Disposal in the Northwest Arabian Gulf Region. 224
M. Al-Haware, P. Al-Masaad and P.S. Joseph.

SESSION III:DREDGED MATERIAL DISPOSAL

Overview of the Cooperative Interagency (CE/EPA) Testing of Dredged Material P rogram-Field Verification Program (FVP). 225
R. Lazor.

Applicability and Field Verification of Predicative Methodologies for Aquatic Dredged Material Disposal 232
J.H. Gentile, G.G. Pesch, K. J. Scott and W. Munns

Field Verification Program Results of Wetland Creation and Upland Disposal Studies. 256
C.R. Lee, J.W. Simmers, B.L. Folsom, Jr., J.G. Skogerboe and M.R. Palermo.

Field Verification Program: Synthesis and Comparative Assessment of Dredged Material Disposal Alternatives. 258
T.M. Dillon.

Page

Evaluation of the Fish Community in Selection of Disposal Sites for Dredged Materials in the Lower Bay Complex of the Hudson-Raritan Estuary. 260
P.M.J. Woodhead.

Reduction and Resolution of Conflicts Between Ocean Dumping and Aquaculture. 262
D. Aggett and A. McIver.

The Tampa Harbour Project: A Major Monitoring Success. 273
J.E. Amson.

An Overview of Dredged Material Ocean Dump Site Monitoring and Assessment Programs and Dump Site Designation Conducted by Environment Canada Under the Ocean Dumping Control Act in the Atlantic Region 292
K.L. Tay.

Simulation of Dredged Material Dispersal During Aquatic Disposal. 316
W.R. Munns, Jr. and J. F. Paul.

Near-Shore Versus Deep-Sea Disposal of Wastes and Dredged Materials Contaminated with Pathogenic Bacteria. 325
D.J. Grimes.

Decision Making Framework for Dredged Material Management. 340
C.R. Lee, R.K. Peddicord and S. Kay

SESSION IV: MANAGEMENT OF WASTES IN THE OCEANS

Wastes in Marine Environments. 341
H. Levenson, K. Wagner, N. Sundt, G. Hund, R. Denison and W. Barnard.

The Role of Sediment Quality Values in Management Decision Making. 348
D.J. Johns, R.C. Barrick and Thomas C. Ginn.

The U.S. EPA Ocean Data Evaluation System. 370
B. Burgan and T. M. Greib.

Page

Models and Observations of Initial Mixing of Ocean-Dumped Wastes: A Status Report. 386
J.P. Downing, R. M. Ecker and M. C. Elbe

A Decision Framework for Site Management and Site Monitoring at Ocean Dumping Sites. 408
M.S. Connor, C.E. Werme, P.D. Boehm and J. M. Neff.

SESSION V:PATHWAYS AND FATES OF WASTES IN MARINE ENVIRONMENTS

Monitoring Release of Trace Metals From Stablilized Oil Ash Reefs. 439
J. H. Trefry, S. Metz, and P.R. Trocine.

A Model for the Dispersion of Radionuclides Released into the Deep Ocean. 441
M. Chartier, X. Durrieu de Madron and M. Poulin.

A Theoretical Look at the Importance of Pollutant Scavenging and Retention in Coastal Environments. 453
K.R. Hinga.

Transfer of Cu, Zn, Pb, As, Fe and Mn in Waste Water/Sea Water Mixed System in Coastal Area of Jinzhou Bay, Bohai Sea. 454
W-y. Qian, D-z. Liu, Y-z. Fu and J-y. Zhou.

Lead Dispersion Gradients in San Francisco Bay. 468
A.R. Flegal and D.R. Smith.

SESSION VI: BIOLOGICAL EFFECTS

Sediment and Biological Conditions in Reference Areas off Southern California, U.S.A. 484
B.E. Thompson, J. Laughlin and D. Tsukada.

Sewage Sludge Disposal: New Approaches to Monitoring for Biological Effects. 487
T. ap Rheinallt, J. Orr and K. Whitelaw.

Concordance Between Laboratory Sediment Bioassays and the Characteristics of Benthic Macroinvertebrate Communities in Commencement Bay, Washington. 489
D.S. Becker, G.R. Bilyard and T.C. Ginn.

Page

The Toxic Effects of Pollutants and Dredged Materials on Marine Organisms. 519
Z. Xu, L. Houng, J. He, X. Zeng and B. Zheng.

The Role of Bacterial Bioassays in Predicting the Potential Impact of Dumping Contaminated Sediments in the Ocean. 540
K. Lee, C. N. Ewing and E. M. Levy.

SESSION VIII: INCINERATION AT SEA

Environmental Acceptability of Ocean Incineration Emissions and Some Remarks on Future Research and Regulations. 562
H. Compaan.

Assessing Potential Effects of Incinerating Hazardous Wastes at Sea: Using Research to Answer Management Questions. 564
D. Redford, M. Jackson, J. Gentile, D. Oberacker, P. Boehm and C. Werme.

Assessing Potential Effects of Incinerating Hazardous Wastes at Sea: Development and Field-Testing of the Marine Biological Assessment Sampler. 573
C. Werme, P. Boehm, M. Cooke, K. Degner, D. Oberacker, M. Jackson and D. Redford.

Proposed Biological Testing Methods for the United States Incineration-At-Sea Research Program. 580
C. J. Strobel, J. H. Gentile, S.C. Schimmel, R.S. Carr and J.W. Williams.

The Changing Characterization of Ocean Incineration Waste Streams and its Impact on the Industry Demand. 594
K. Haubourdin and G. P. Nassos.

A New Method for Collecting, Transporting and Incinerating Aboard Ship -- Toxic Materials and Hazardous Wastes. 613
V. G. Grey.

Page

The Environmental Safety and Acceptability of Incineration at Sea. 620
J. Karau.

SESSION VIII: RISK ASSESSMENT IN MARINE WASTE MANAGEMENT

Ecological and Human Health Risks for Sewage Sludge Disposal at the 106-Mile Site. 625
H. A. Walker, J. F. Paul., J. A. Nocito-Gobel and J. H. Gentile.

Comparative Risk Assessment for Marine Disposal of Sewage Sludge at Midshelf and Offshelf Sites. 644
J.A. Nocito, H. A. Walker, J. F. Paul and C.A. Menzie.

Preferences for Near-Shore Versus Deep-Sea Disposal of Wastes. 664
T. M. Leschine, W.A. Verdini and H. Wallenius.

Risk Assessment for Subseabed Disposal of High Level Radioactive Waste. 666
R. D. Klett.

BANQUET SPEECH
Justifying Sea Disposal of Wastes. 689
R. G. Boelens.

APPENDIX A: EXECUTIVE AND LOCAL COMMITTEE MEMBERSHIP A-1

APPENDIX B:LIST OF PARTICIPANTS B-1

ROLE OF OCEAN DISPOSAL IN CANADA'S WASTE MANAGEMENT STRATEGY

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Ottawa, Ontario
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ABSTRACT

Canada recognizes that a high standard of marine environmental quality is essential to protect and sustain; fisheries, specific industries, traditional life styles, tourism and recreation, and diverse biota, habitats and ecosystems. Poor marine environmental quality contributes to the high cost of pollution controls and clean up, less international consumer confidence in Canadian fish products, closed fisheries due to contamination, less tourism in contaminated areas, and degraded habitats for natural populations of commercial species of fish, seabirds and marine mammals.

The objectives of both the London Dumping Convention (LDC) and our Ocean Dumping Control Act (ODCA) pledge Canada to take effective measures to prevent marine pollution by dumping and to promote control of all sources of marine pollution. Ocean dumping of wastes into the oceans forms a minor source of marine pollution relative to land-based discharges, air-borne pollutants and river inputs; however, the marine pollution problems faced are similar and the London Dumping Convention provides a key international role in promoting marine environmental quality.

The following waste management principles are taken into account whenever the sea disposal option is considered:

1. wherever possible, recycle and reuse waste products;
2. wastes that cannot be recycled or reused should be treated at the source to the extent possible;
3. wastes that cannot be avoided or reused at a reasonable cost must be disposed of safely; and
4. sea disposal should only be used if it poses less or no larger human health and environmental risks than practicable land-based alternatives.

Approximately 75 percent of Canada's ocean dumping permits are issued for the sea disposal of dredged material. Non-dredged material permits generally cover such activities as oil spill experiments, incineration of galley wastes, disposal of decommissioned vessels, scrap metal, construction rubble, fish offal, firearms and munitions, experiments involving small quantities of radioactive material, and the use of surfactants to accelerate the freezing of seawater.

BACKGROUND

Canada recognizes that a high standard of marine environmental quality is essential to protect and sustain; fisheries, specific industries, traditional life styles, tourism and recreation, and diverse biota, habitats and ecosystems. We view the oceans as the common heritage of mankind. A moral need exists to protect for existing and future generations the marine environment and ensure that its quality and resources are not impaired.

The objectives of both the London Dumping Convention (LDC) and our Ocean Dumping Control Act (ODCA) pledge Canada to take effective measures to prevent marine pollution by dumping and to promote control of all sources of marine pollution. Ocean dumping of wastes into the oceans forms a minor source of marine pollution relative to land-based discharges, air-borne pollutants and river inputs; however, the marine pollution problems faced are similar and the London Dumping Convention provides a key international role in promoting marine environmental quality.

WASTE MANAGEMENT STRATEGY

We endorse the application of environmentally responsible technology including waste treatment at the source, rather than the simple selection of the least cost waste disposal alternative. Waste treatment must be viewed as a legitimate cost of doing business and the control requirements must, by definition, be flexible enough to provide the necessary consideration for both technical feasibility and economic viability.

The ODCA calls for a comprehensive waste management approach which promotes the effective control of all sources of pollution and requires a comparative assessment of the land and sea disposal options. Dumping at sea is permitted only in cases where the disposal of wastes meets the ODCA regulatory requirements and no other environmentally preferable and practical alternative is available. If ocean dumping is not permitted, we will work with the appropriate federal and provincial agencies to investigate alternative disposal methods.

The following waste management principles are taken into account whenever the sea disposal option is considered:

1. wherever possible, recycle and reuse waste products;
2. wastes that cannot be recycled or reused should be treated at the source to the extent possible;
3. wastes that cannot be avoided or reused at a reasonable cost must be disposed of safely; and
4. sea disposal should only be used if it poses less or no larger human health and environmental risks than practicable land-based alternatives.

In applying principle 4, alternatives to ocean disposal are considered in the light of a comparative assessment of:

- Human health risks;
- Environmental costs;
- Hazards (including accidents) associated with treatment, packing, transport and disposal;
- Economics (including energy costs);
- Exclusion of future uses of disposal areas

for both sea disposal and the alternatives.

If the foregoing analysis shows that ocean disposal poses less or no larger human health and environmental riks, ocean disposal is permitted.

CANADIAN POLICIES

Canada is primarily a non-dumping nation for hazardous wastes. Ocean dumping of waste will be permitted if wastes that cannot be avoided, recycled or reused at a reasonable cost can be safely disposed of at sea.

Environment Canada has examined the available information on incineration at sea and has found no data to indicate that it is an unacceptable method of waste disposal. We are therefore prepared to consider issuing an ocean dumping permit for incineration at sea if an applicant can meet the regulatory requirements (LDC regulations and ODCA permit conditions) and there are no practical alternatives that are environmentally preferable.

Canada is a "non-dumping" country for radioactive wastes, and has no plans to dump such wastes at sea. We support the voluntary LDC moratorium on this practice and are holding in abeyance any permit decisions pending the outcome of the results from the ongoing LDC review of this issue.

For the import/export of wastes for disposal at sea, Canada would require an ODCA loading permit for export and a LDC dumping permit, or similar approval, before allowing export for dumping at sea.

WASTES DUMPED AT SEA

Approximately 75 percent of Canada's ocean dumping permits are issued for the sea disposal of dredged material. Non-dredged material permits generally cover such activities as oil spill experiments, incineration of galley wastes, disposal of decommissioned vessels, scrap metal, construction rubble, fish offal, firearms and munitions, experiments involving small quantities of radioactive material, and the use of surfactants to accelerate the freezing of seawater. Table 1 provides an overview of the ocean dumping permits issued and used in 1985. In addition, eleven permit applications were withdrawn or rejected. Reasons for rejection or withdrawal of permit applications include the identification of prohibited and restricted substances in the material to be dumped, and the cancellation of projects.

FORECAST

Population growth and industrial expansion will lead to increasing pressure on the sea as a source of living and non-living resources, as a source of energy, as a medium of transportation, and as an ultimate respository for all kinds of wastes.

Dredging operations are expected to follow current trends in Canada. Approximately 90% of the dredged material dumped at sea is considered relatively innocuous and the primary environmental concern is the direct physical impact of dumping. Conflicts do arise between selecting sites for aquaculture and dumping dredged material. Dredged material taken from industrialized areas is usually contaminated with heavy metals, synthetic organics, oil and grease and requires special care for disposal.

Finding an appropriate means of disposal for accumulated solid waste (e.g., fuel drums, old appliances, electrical equipment, vehicles, and machinery) continues to be a problem in northern communities, at abandoned military sites, and at abandoned fuel cache sites. In addition, the volume of solid wastes being produced along the arctic coast is increasing in areas where industrial development is underway (e.g., the Beaufort Sea). Serious limitations exist on the number and availability of acceptable landfill sites, the waste suitability for incineration, and the expense of recovery or recycling. Sea dumping of metal wastes has been initiated and the practice is expected to continue.

In the Pacific Region, the number and availability of acceptable landfill sites for industrial waste (e.g., spent pot lining from an aluminum smelter and

Table 1: 1985 Ocean Dumping Permits

STATISTICS: QUANTITIES APPROVED

Dredged Material	Total Quantity	No. of Permits	Percent of Permits
Dredged Material (density = 1.4)	8 032 932.6 tonnes (5 737 809.0 m^3)	97	75.2%
Vessels	5 181.9 tonnes	12	9.3%
Offal	37 936.0 tonnes	9	7.0%
Oil Experiments (density = 0.86)	2.2 tonnes (2 590 litres)	2	1.6%
Scrap Metal	105.0 tonnes	2	1.6%
Brine Solution (density = 1.197)	60 448.5 tonnes (50 500.0 m^3)	2	1.6%
Surfactant AFA-6 (density = 1.4)	2.8 tonnes (2 000.0 litres)	2	1.6%
Experiments with Radioactive Material	9.25×10^8 Bq	1	0.7%
Firearms and Prohibited Weapons	500.0 pieces	1	0.7%
Ship Galley Refuse	1 000.0 tonnes	1	0.7%
TOTAL	~8×10^6 tonnes	129	100%

construction debris) is rapidly decreasing, and marine disposal is currently being investigated.

In the Atlantic Region, decreasing markets for fish offal may lead to a heavier reliance on ocean disposal. The situation is likely to remain stable in Newfoundland where most of the fish offal is ocean dumped because there is no recycling market. In Nova Scotia and New Brunswick, ocean disposal is seen as a potentially attractive alternative to fish meal plants and land filling. As a result, there is economic pressure for designation of fish offal ocean dump sites closer to shore. The quantities proposed for dumping will continue to be linked to the success of herring catches and the availability of disposal alternatives.

The interest in ocean disposal of sewage sludge is increasing. A permit was issued and used on the Pacific Coast, for 136 500 litres of domestic sewage sludge. On the East Coast, the direct discharge of sewage sludge into Halifax harbour has been a general public concern. The Municipality of Halifax is presently involved in a study to assess the implications of sewage sludge disposal methods, and ocean dumping is one of the possible options.

The increasing need for waste destruction compared to the availability of land-based options maintains the need for incineration at sea as a method of waste disposal.

CONCLUSION

IN CONCLUSION, the Canadian view is that the ultimate goal of the LDC & ODCA should be the elimination of dumping all hazardous wastes posing an unacceptable risk, the best possible and practicable control of wastes that are ocean dumped and the encouragement of a more holistic environmental consideration of the acceptability of any proposed disposal technique.

We also believe that when considering the role of ocean disposal on a global scale, the LDC must address the political, economic or timing constraints of individual Contracting Parties, but nevertheless accept as a consistent basis for any discussion an approach based upon science and technology.

REFERENCES

Environment Canada. Keeping the Ocean Clean. Ocean Dumping Control Act 1985/86 Annual Report. 34 pp.

STABILIZED INCINERATION RESIDUE: A POTENTIAL MATERIAL FOR ARTIFICIAL REEF CONSTRUCTION

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ABSTRACT

With the closure of Long Island landfills by 1990 the majority of communities are considering incineration of the huge amounts of refuse generated daily in this region. An important obstacle to the acceptance of incineration plants is the large volume of combustion products, fly ash, bottom ash and dry scrubber residue, which require disposal. The problem is particularly difficult in urban areas near the coast where potential disposal sites are few and increasingly distant.

In May 1985 a research program began to examine the feasibility of stabilizing incineration wastes. Residue from three operational mass burn facilities was collected and fixated using Portland cement. The stabilization process formed a solid block possessing physical properties appropriate for marine disposal. The stabilized material was subjected to EP, TCLP and other extraction protocols and in no instance did the metal concentrations in the resultant leachates exceed regulatory limits for toxicity. Bioassays reveal no adverse impacts to the phytoplankton species exposed to elutriate concentrations significantly higher than could be encountered under normal disposal conditions. The success of our laboratory investigations resulted in securing the necessary permits and the placement of a small number of stabilized incineration ash blocks to simulate an artificial habitat in Conscience Bay, Long Island Sound.

In the months ahead, divers will return to the site to record the physical, chemical and biological interactions of these blocks with the marine environment.

INTRODUCTION

There is an urgency in the New York Metropolitan area to convert from landfilling to incineration of the millions of tons of garbage and trash produced annually. An important obstacle to the acceptance of incineration plants is the large volumes of combustion products, fly ash and bottom ash, which require disposal. The problem is particularly difficult in urban areas near the coast where potential disposal sites are few and increasingly distant. In addition, the potential for groundwater pollution from leachates produced at an ashfill site make landfilling of incineration wastes a less than popular choice.

A research program was begun in May 1985 at the Waste Management Institute of the Marine Sciences Research Center (MSRC) at the State University of New York at Stony Brook to examine the feasibility of stabilizing incineration wastes. The primary objective of this project is to fabricate solid blocks made of incineration wastes and examine these blocks for their physical and chemical properties with the eventual placement of such material in the sea. This investigation has produced information which shows that incineration wastes possess excellent characteristics for stabilization.

The project, though still in its early stages of development, has achieved considerable success. Results of fabricating a variety of different mixes of incineration

wastes with small amounts of stabilizing additives, lime and Portland cement, have produced solid proctor samples having excellent structural properties. Because of their hardness, incineration wastes blocks may have an important and novel use in underwater construction, particularly the building of artificial fishing reefs and in other marine applications such as submerged breakwaters. Except for the preliminary investigations conducted at MSRC, there are at present virtually no reported data on the behavior of stabilized incineration ash (SIA) blocks in the sea.

PRIMARY OBJECTIVES

The project focused on four main objectives:

OBJECTIVE 1: To investigate incineration ash fixation by means of chemical additives and controlled curing environments.

OBJECTIVE 2: To determine the physical and chemical characteristics of several "optimum" stabilized mixes.

OBJECTIVE 3: To evaluate the potential impact to the marine environment from the disposal of stabilized incineration residue.

OBJECTIVE 4: To establish an artificial habitat in the coastal waters off Long Island and monitor the interactions of stabilized incineration residue with the marine environment.

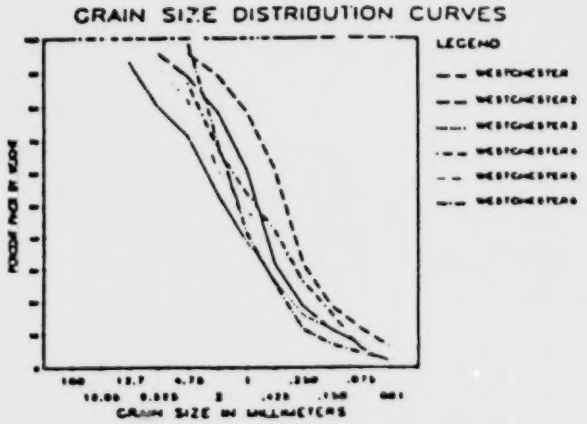
PHASE 1. OPTIMUM MIX FABRICATION

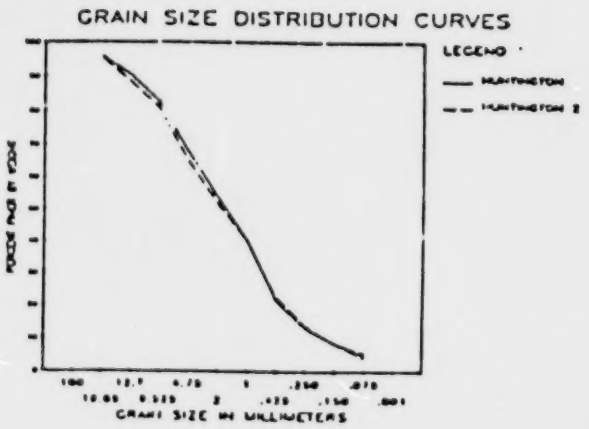
Incineration ash was s[illegible]ured from three operational incinerators. New York City's Southwest Brooklyn facility twice provided a source of fly ash. On two separate occasions the Town of Huntington incinerator located in East Northport, Long Island was visited to remove composite samples of both bottom and fly ash. Composite ash samples were also acquired from the Westchester Resource Recovery facility in Peekskill, New York.

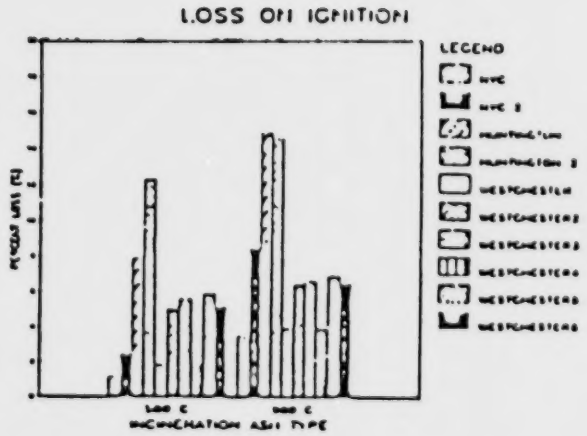
Incineration wastes were initially characterized by determining the moisture content of the residue, particle size distribution of the residue and the pH of incineration residue-distilled-deionized water mixtures. Grain size distribution curves show that incineration residues are predominantly in the sand size range (<4.75 mm, >0.075 mm) with lesser amounts in the gravel (>4.75 mm) and silt or clay sizes (<0.075 mm) (Figure 1). Moisture content of the residues ranged from 2-27%, and the pH of incineration ash-distilled-deionized water mixtures ranged from 8.8-12.7.

Loss on ignition at 500°C and 900°C was used to estimate the organic content of incineration residues (Figure 1). X-ray diffraction analysis identified anhydrite, gypsum, quartz, calcite and ettringite as major crystalline phases in incineration residue.

Incineration wastes were combined with Portland cement and water to form proctor sized cylinders (Table 1). The ASTM "Standard Test Moisture-Density Relations of Soils and







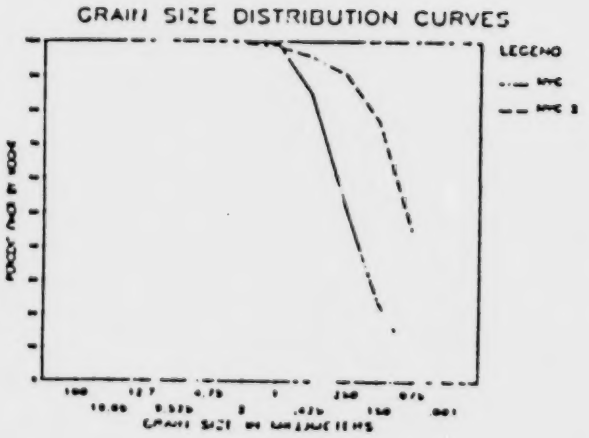


Figure 1. Grain Size Distribution curves and loss on ignition for incineration residues.

Table 1. Summary of physical characteristics of optimum mixes.

Test	New York City Fly Ash (COB)	Huntington Composite Residue (HOB)	Westchester Composite Residue (WOB)
Proctor fabrication data (Modified ASTM D698 method)	62% Ash 15% Portland Cement 23% Moisture	67% Residue 15% Portland Cement 18% Moisture	68% Residue 15% Portland Cement 17% Moisture
Proctor compressive strength (psi)	228 (±43)	455 (±51)	1230 (±59)
Proctor wet density, ASTM D698 (modified) (kg/m^3)	1530	1710	1740
Darcy falling head permeability (cm/sec)	Top of Proctor: 10^{-7} Bottom of Proctor: 10^{-5}	T: $<10^{-7}$ B: $<10^{-5}$	T: 10^{-8} B: 10^{-8}
Density/porosity, ASTM C642:			
Apparent specific gravity	2.23	2.53	2.51
Water absorption	36%	21%	18%
Voids	47%	39%	36%

Soil-Aggregate Mixtures using 5.5 lb Rammer and 12 inch Drop" procedure was used for proctor fabrication (ASTM, 1975). While modifications of this method were investigated, proctor fabrication could not be improved.

A water content of 17% was found to be optimum for fabrication of proctors containing Westchester residue (Table 1). In general optimum water content is unique for each mix design and is a function of median particle size, with water content increasing as particle size decreases.

The maximum observed compressive strength was 1590 psi for a 67% Westchester residue and 15% Portland cement mix with 18% water content cured for 21 days at ambient temperatures (25°C). Additions of sodium carbonate (Na_2CO_3), found to improve the structural integrity in fixated coal ash, resulted in no improvement for the incineration ash products. Gypsum, calcium sulfate dihydrate ($CaSO_4.2H_2O$), another commonly used additive, significantly deteriorated the compressive strength resulting in blocks with the lowest compressive strengths measured during this investigation.

Based on the results of compressive strength testing of numerous incinerator ash proctors, three optimum mixes were selected for additional physical and chemical testing;

- 62% New York City fly ash with 15% Portland cement and 23% moisture cured at 49°C for 72 hours. [Proctor series COB].

- 67% Huntington combined residue with 15% Portland cement and 18% moisture cured for 14 days at ambient temperatures. [Proctor series HOB].
- 68% Westchester combined residue with 15% Portland cement and 17% moisture cured at 49°C for 24 hours. [Proctor series WOB].

An in depth account of the stabilization program is presented in Roethel et al., 1986.

PHASE 2. PHYSICAL AND CHEMICAL CHARACTERIZATION OF THE OPTIMUM MIXES

The next stage of the project addressed the second objective; to determine the permeability and leachate characteristics of the optimum mixes. Portions of proctors were sealed into PVC pipe. The rate and quantity of water which permeated through the proctors was recorded, enabling calculations of permeability constants. In general, the permeabilities of the stabilized incineration residue proctors were less than 10^{-7} cm/s. A summary of the physical characteristics of the optimum mixes is shown in Table 1.

The elemental composition of the stabilized incineration blocks was investigated. For comparison purposes the elemental composition of a cement block was also determined. Samples of the stabilized blocks were ground to a powder and were digested in hydrofluoric-boric acid prior to metal analysis of the digests by atomic absorption spectrophotometry (AAS).

Major constituents common to all three stabilized blocks were Al, Mg, Fe, Mn, and Ca (Table 2). Copper, Cr, Zn, Mn and Pb, elements enriched in particulate incineration residues (Roethel et al., 1987), were observed in minor concentrations in the stabilized blocks. Cadmium, Hg, Co, Ni and As were found in trace amounts.

An extensive evaluation of the leachate characteristics of stabilized incineration ash blocks was undertaken. Leachate samples were prepared using EPA and ASTM procedures and tested for fifteen elements by AAS. The EPA Toxicant Extraction Procedure (EP), (U.S. EPA, 1980) and the EPA Toxicant Characteristic Leaching Procedure (TCLP) (U.S. EPA, 1986) along with the ASTM Leaching Procedure (ASTM, 1980) using both distilled water and filtered seawater were performed on crushed samples to evaluate their leaching behavior in acidic, neutral and alkaline solutions. Concentrations for six elements measured in the four leachates are presented in Table 3. Throughout our extensive testing program metal concentrations in leachates from stabilized incineration residues did not exceed EPA limits for toxicity. A complete data set of the leachate investigations can be found in Roethel et al. (1987).

PHASE 3. MARINE DISPOSAL IMPACTS

Blocks placed in seatables at the Flax Pond Laboratory, Long Island Sound were removed periodically during the year to evaluate changes in the materials structural integrity following submersion in the sea. Blocks of Westchester and

Table 2 Metal concentrations in stabilized incineration ash blocks.

	Ca (%)	Mg (%)	Al (%)	Fe (%)	Cu (%)	Pb (%)	Zn (%)	Cd (lg/g)	Mn (%)	Ni (lg/g)	Cr (lg/g)	Co (lg/g)	As (lg/g)	Hg (lg/g)
CB[a]	6.6	1.9	5.4	4.5	0.0013	0.0004	0.0053	0.26	0.092	47	31	25	33	BDL[d]
SD[b]	0.8	0.3	0.2	0.9	0.00001	0.0002	0.0003	0.10	0.015	3	7	5	5	
WOB[c]	14	1.4	5.8	4.9	0.47	0.51	0.45	44	0.083	110	150	20	4.8	11
	0.5	0.1	0.6	0.4	0.26	0.21	0.03	5	0.001	10	4	2	0.8	6
HOB	13	0.96	4.0	7.3	0.14	0.10	0.23	13	0.081	55	140	12	16	BDL
	0.3	0.10	0.2	0.2	0.04	0.03	0.06	2	0.008	6	10	1	2	
COB	13	1.3	7.0	1.6	0.061	0.73	1.86	730	0.11	78	190	20	73	1.3
	0.4	0.0	0.1	0.1	0.001	0.03	0.01	30	0.0003	13	3	1	2	0.9

[a] Paricle size used: <9.5 mm for CB, WOB, HOB and COB.

[b] Values denote the standard deviation of replicate samples (n=3).

[c] Optimum mix designs are shown in Table 1.

[d] BDL means below detection limit; 1 (lg L^{-1}) for Hg.

Table 3. Metal concentrations in leachates from stabilized incineration ash blocks.

	Cu	Pb	Zn	Cd	As	Hg
Regulatory Limits	-	5.0	-	1.0	5.0	0.2
ASTM:						
WOB	0.3	0.6	0.2	<0.001	<0.001	<0.002
HOB	1.6	0.04	0.05	*	0.006	*
COB	0.09	0.5	1.2	*	0.005	*
SS:						
WOB	0.2	<0.005	-	*	*	*
HOB	2.7	*		*	0.002	*
COB	0.005	0.01		0.005	0.006	*
EP:						
WOB	0.14	<0.001	0.05	*	*	*
HOB	0.92	*	0.04	*	0.005	*
COB	<0.001	0.003	0.054	0.11	0.002	*
TCLP:						
WOB	0.13	*	0.04	0.002	*	*
HOB	0.85	*	0.03	*	0.005	*
COB	*	0.02	0.24	0.35	0.002	*

Unit of concentration is Mg L^{-1}
* Denotes below detection limit
0.05 (μg/l) for Cd and As
0.5 (μg/l) for Pb
2 (μg/l) for Hg

Huntington residue maintained their structural integrity following submersion while the blocks fabricated using New York City fly ash crumbled after 125 days of submersion (Figure 2). Anhydrite, a major component of NYC fly ash, expands following submersion in seawater to form gypsum which accounts for the loss of structural integrity.

Bioassay experiments were designed to determine toxic potentials, if any, of soluble components from stabilized incineration waste blocks in seawater, upon representative marine plants, - phytoplankton. The bioassay basically measures the growth responses of the plant cultures to the product of toxicant concentration and exposure time. Therefore, cultures of plants with high growth rates and short generation times are more likely to show measurable responses. The diatom Thalassiosira pseudonana was chosen for our assays because it occurs commonly in New York inshore marine environments, it has a short generation time (doubling daily in laboratory cultures) and it has been recommended by EPA as a species sensitive to metals. Our laboratory has used this species in standard bioassays for toxicity of a number of materials in seawater.

The bioassay experiments were designed to determine toxic or sub-lethal effects of seawater leachates from incinerator waste blocks on marine phytoplankton growth and biomass, in culture. A one-way analysis of variance (ANOVA) was used to test the statistical significance of the

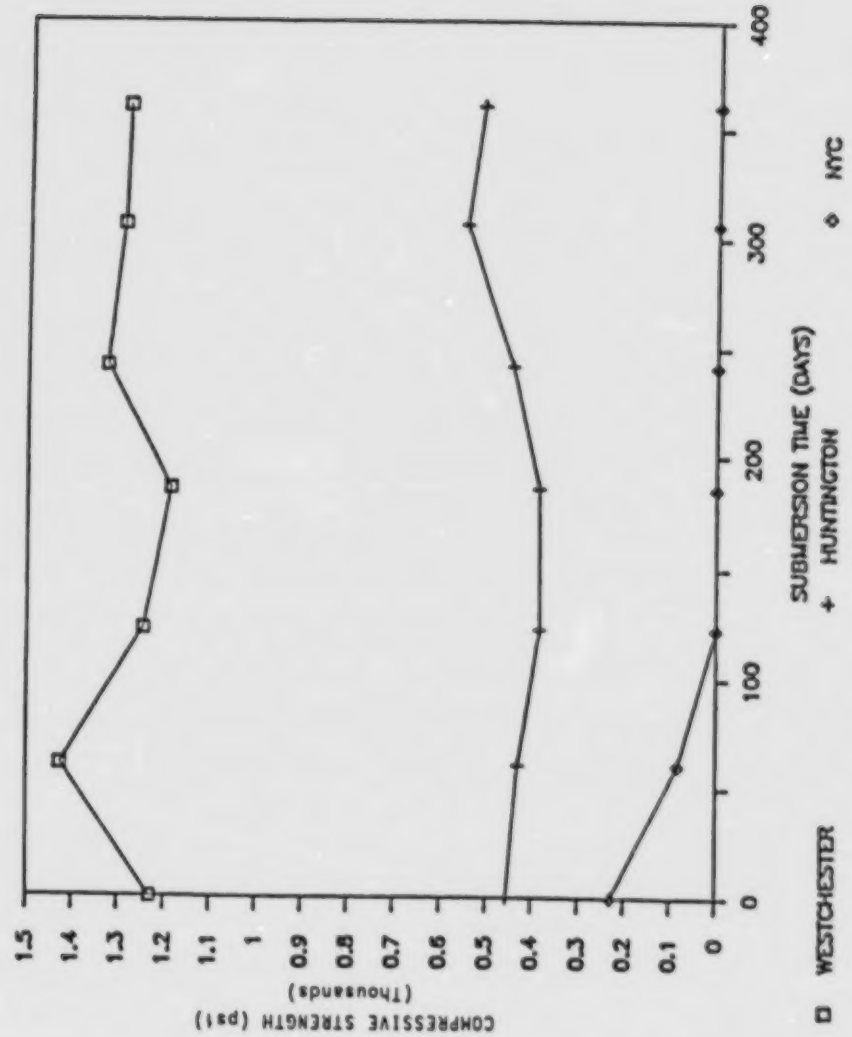


Figure 2. COMPRESSIVE STRENGTH vs SUBMERSION TIME

differences between the control means and the leachate treatment means using a probability of $p < 0.025$.

Results of the assays with T. pseudonana cultures on leachates from two separate batches of stabilized Westchester wastes, collected in August 1985 and January 1986, are given in Tables 4 and 5. The first batch of materials showed no significant effects ($p < 0.025$) of treatment with any concentration of leachate until the third day of assays, and then chlorophyll a pigment (biomass) and ^{14}C uptake (photosynthesis) were significantly depressed only by the highest, 10% concentration of leachate. No effects were detected on the third day with lower concentrations of incinerator waste leachate in cultures. The leachate from the second batch showed significant inhibitions ($p < 0.025$) for ^{14}C uptake on day 1, and for both chlorophyll a and for ^{14}C uptake on day 2, - but inhibition occurred only at the highest, 10%, concentration of leachate. No significant inhibitions were detected at lower concentrations of leachate on any day of the bioassay. It should be noted that there was a significant ($p < 0.025$) stimulus of phytoplankton biomass (measured as chlorophyll a) on day 1 in cultures with waste block leachate at 1.0% and 10% concentrations. Such stimulation is not uncommon in bioassays and may be due to the presence of small amounts of plant growth stimulants in the leachate.

Table 4. Bioassay with Thalassiosira, Westchester MSW, batch 1.

		Control	0.1%	1.0%	10.0%
Day 0	Chl-a	2.64	2.23	2.13	2.11
	C-14	3.98	4.12	3.93	4.58
	Cell/ml	21.1×10^3	14.2×10^3	13.1×10^3	11.3×10^3
Day 1	Chl-a	16.65	15.66 ns	15.05 ns	16.03 ns
	C-14	7.26	7.75 ns	10.30 00	7.83 ns
	Cell/ml	25.7×10^3	32.4×10^3 ns	29.9×10^3 ns	28.6×10^3 ns
Day 2	Chl-a	80.80	85.20 ns	92.80 ns	76.00 ns
	C-14	90.47	77.23 ns	87.30 ns	55.80 ns
	Cell/ml	11.7×10^4	14.2×10^4 ns	9.5×10^4 ns	10.3×10^4 ns
Day 3	Chl-a	396.2	386.1 ns	383.5 ns	338.0 **
	C-14	353.5	319.9 ns	342.3 ns	243.1 **
	Cell/ml	68.6×10^4	67.8×10^4 ns	70.6×10^4 ns	62.5×10^4 ns

** Indicates a significant decrease in mean ($p < 0.025$).

ns Indicates no significant difference ($p < 0.025$).

00 Indicates significant increase ($p < 0.025$).

Chl-a is chlorophyll a content, μg liter^{-1}.

C-14 is uptake of labelled carbon, ^{14}C, cpm ml^{-1}.

Cell/ml is cell volume, μl ml^{-1}.

Table 5. Bioassay with Thalassiosira, Westchester MSW, batch 2.

		Control	0.1%	1.0%	10.0%
Day 0	Chl-a	1.76	1.71	2.27	2.15
	C-14	3.07	2.82	3.90	1.72
	Cell/ml	6.99×10^3	5.98×10^3	7.87×10^3	7.44×10^3
Day 1	Chl-a	6.28	6.08 ns	7.96 00	6.72 ns
	C-14	5.43	4.96 ns	5.47 ns	3.85 **
	Cell/ml	24.8×10^3	22.8×10^3 ns	23.6×10^3 ns	26.5×10^3 ns
Day 2	Chl-a	35.0	32.8 ns	38.9 ns	27.3 ns
	C-14	26.8	23.3 ns	25.51 ns	16.69 **
	Cell/ml	5.40×10^4	5.24×10^4 ns	5.88×10^4 ns	3.55×10^4 ns
Day 3	Chl-a	143.0	140.4 ns	166.4 ns	100.8 ns
	C-14	107.1	97.0 ns	115.3 ns	67.4 ns
	Cell/ml	30.0×10^4	24.4×10^4 ns	42.5×10^4 ns	24.0×10^4 ns

** Indicates a significant decrease ($p < 0.025$).

ns Indicates no significant difference ($p < 0.025$).

00 Indicates significant increase ($p < 0.025$).

Chl-a is chlorophyll a content, μg liter^{-1}.

C-14 is uptake of labelled carbon, ^{14}C, cpm ml^{-1}.

Cell/ml is cell volume, μl ml^{-1}.

PHASE 4. PLACEMENT AND MONITORING OF A MARINE ARTIFICIAL HABITAT

BLOCK FABRICATION

During the period December 8-12 1986, experiments in incineration ash block fabrication were made at the research facilities of the Besser Company sited at the Alpena Community College, Alpena, Michigan. Principal aims of these investigations were to manufacture blocks of different cement and incineration ash mix designs employing conventional machines currently used by commercial concrete block factories, and to investigate proper conditions for accelerated curing of incineration waste blocks in the steam kiln. These investigations were designed to produce both artificial reef quality blocks and construction grade blocks.

Two ash types were collected from the Westchester incinerator in November, 1986. These ash types were bottom ash only and the normal output of mixed bottom ash and fly ash (referred to as combined ash). The reason for this separation of the ash was to observe which ash type would be more easily stabilized. Processing of this ash at the Alpena facility prior to block fabrication involved screening the ash to obtain particle sizes less than 3/8". The material larger than this sieve size was crushed with a jaw-crusher and then resieved. It was noted that the material remaining on the sieve contained larger amounts of metal than previously sieved ash samples. The metal

fragments were found after crushing large glass fragments. The glassious part of these chunks would shatter into fragments smaller than the sieve with the larger metal pieces retained on the sieve. Thus, using the jaw-crusher and screening has the beneficial effects of removing more of the metal present while increasing the amount of silica (from the crushed glass and rocks), thus improving the characteristics of the ash prior to block making.

Block production proceeded without any unusual problems. Blocks were steam cured for 24 hours and moved on pallets by forklift to the yard where the curing process continued. The compressive strength of the blocks following initial steam curing were excellent. Compressive strengths in excess of 1000 psi were measured. Previous studies (CWARP, 1979; Roethel, 1983) have shown that approximately 300 psi is necessary for blocks destined for marine disposal.

CONSCIENCE BAY HABITAT

On April 27, 1987 thirty blocks of stabilized incineration ash (SIA) fabricated during our visit to the Besser Company were submerged in 30 ft. of water in Conscience Bay, Long Island Sound (Figure 3). For comparison purposes, thirty standard cement blocks were placed within 10 ft. of the SIA structure. Within one month following placement, divers returned to the sight and observed hydroids attached to all sides of both structures. Blackfish, cunner and winter flounder were commonly observed

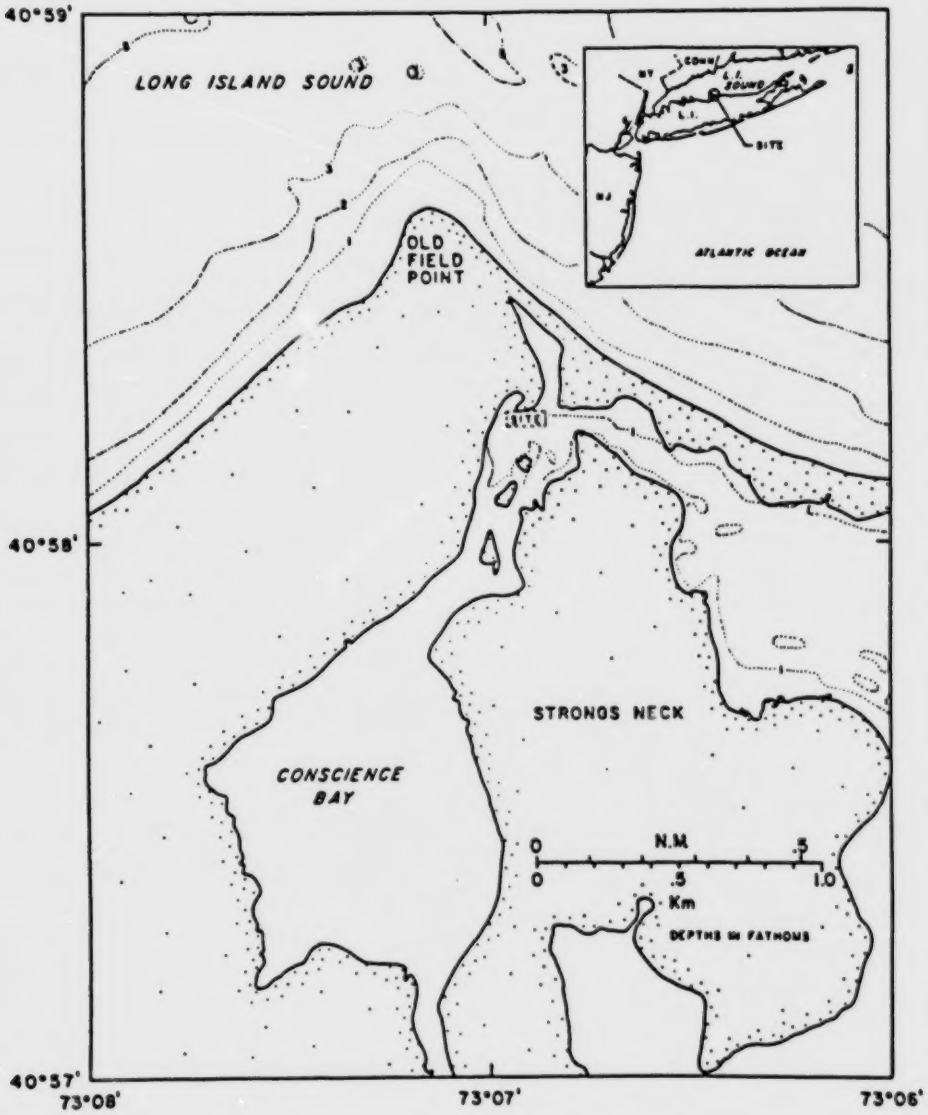


Figure 3. Conscience Bay, Long Island Sound.

in and around the site. By early July, hydroid colonies still represented the dominant fauna, however red and green algae were observed on both structures. Blocks were removed at this time for physical and chemical testing and compressive strength measurements reveal an increase in the structural integrity of the stabilized incineration ash blocks had occurred following submersion in the sea. In the months ahead experiments designed to evaluate potential uptake of block constituents by the attached biological community will be initiated. In addition, alterations in the physical and chemical characteristics of the blocks due to seawater exposure will be investigated.

CONCLUSIONS

1. Stabilization of incineration residue prevents the rapid mobilization and solubilization of the major, minor and trace elements present in the ash.
2. Results of metal analysis of SIA leachates suggest that heavy metal contamination by SIA in the sea is not likely to be environmentally significant in aerobic environments.
3. Under laboratory conditions, elutriates made from SIA seawater mixtures did not adversely effect the productivity of marine algae.
4. Selected mixes of SIA blocks were shown to maintain their structural integrity for extended periods of time in tanks of flowing seawater.

5. Short term in situ data also indicates improved structural integrity following submersion.

6. Early colonization of in situ SIA and concrete blocks began soon after placement in the sea.

ACKNOWLEDGEMENTS

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SHORT-TERM EFFECTS OF STABILIZED OIL ASH REEFS ON THE MARINE BENTHOS

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An experimental artificial reef array located 2 km off the central Florida east coast was installed on April 7, 1987 to examine the suitablity of using stabilized oil ash as a material for reef construction. Biotic components examined to determine whether oil ash reefs differed from control concrete reefs were the infaunal benthos around the reefs, the epifaunal fouling communities on the reefs, and the fishes inhabiting the reefs.

This artificial reef work is unusual in that biological effects were evaluated by use of an experimental design which established three treatments (stabilized oil ash reefs, concrete control reefs, benthic control areas), each replicated two times, in a randomized complete blocks design.

Analysis of settlement of the barnacle Balanus venustus over a four month period showed no significant differences in settlement density depending on reef type. Abundance of this species declined and mortality increased over time on both reef types. These changes may be the results of fish grazing.

Recruitment of fish was rapid, with fishes being observed within 2 d of installation. By four months after installation, mean number of fish per reef had risen to 244 for the oil-ash reefs and 262.5 for the concrete reefs. There was no statistically significant difference in abundance of fish between reef type on any sample date. Composition of fishes on all reefs was similar, with two species, the pigfish (Orthopristis chrysoptera) and the spadefish (Chaetodipterus faber), dominating in abundance.

Benthic infaunal samples taken three months after reef installation showed no significant differences in either abundance or number of taxa between the two reef types. Both paramters were significantly lower within 1 m of both reef types as compared to greater distances. This halo effect may be a result of the increase in numbers of benthic feeding fishes recruiting to the reefs.

There appear to be no differences in any component of the benthic fauna due to reef composition in the short-term.

Introduction

The burning of fuel oil in electric power generation produces an ash residue which is difficult to dispose of in traditional on-land disposal sites. Stabilization of the oil-ash material by mixing it with concrete to form stabilized oil-ash blocks provides a potential marine disposal option in the form of the construction of artificial reefs. Use of blocks composed of stabilized coal-fly ash have been successfully used in the construction of experimental artificial reefs (Parker et al., 1981, 1982; Woodhead and Jacobson, 1985; Humphries et al. 1985). However, the use of oil-ash material presents a potential problem in that oil-ash typically is elevated in concentrations of nickel, copper, cobalt, arsenic and particularly vanadium. Vanadium concentrations in the stabilized oil-ash blocks are typically on the order of 1-2% (Trefry, Metz and Trocine, 1987).

An experimental artificial reef array located 2 km off the central Florida east coast was installed on April 7, 1987 to examine the potential for using stabilized oil-ash as a material for reef construction. The first phase of biological research is designed to evaluate whether biological communites associated with stabilized oil ash reefs differ in any way from control situations over a short-term period. Biotic components which were examined over the initial 4 month period of installation were the epifaunal fouling communities on the reefs, the fishes utilizing the reefs, and the infaunal benthos around the reefs.

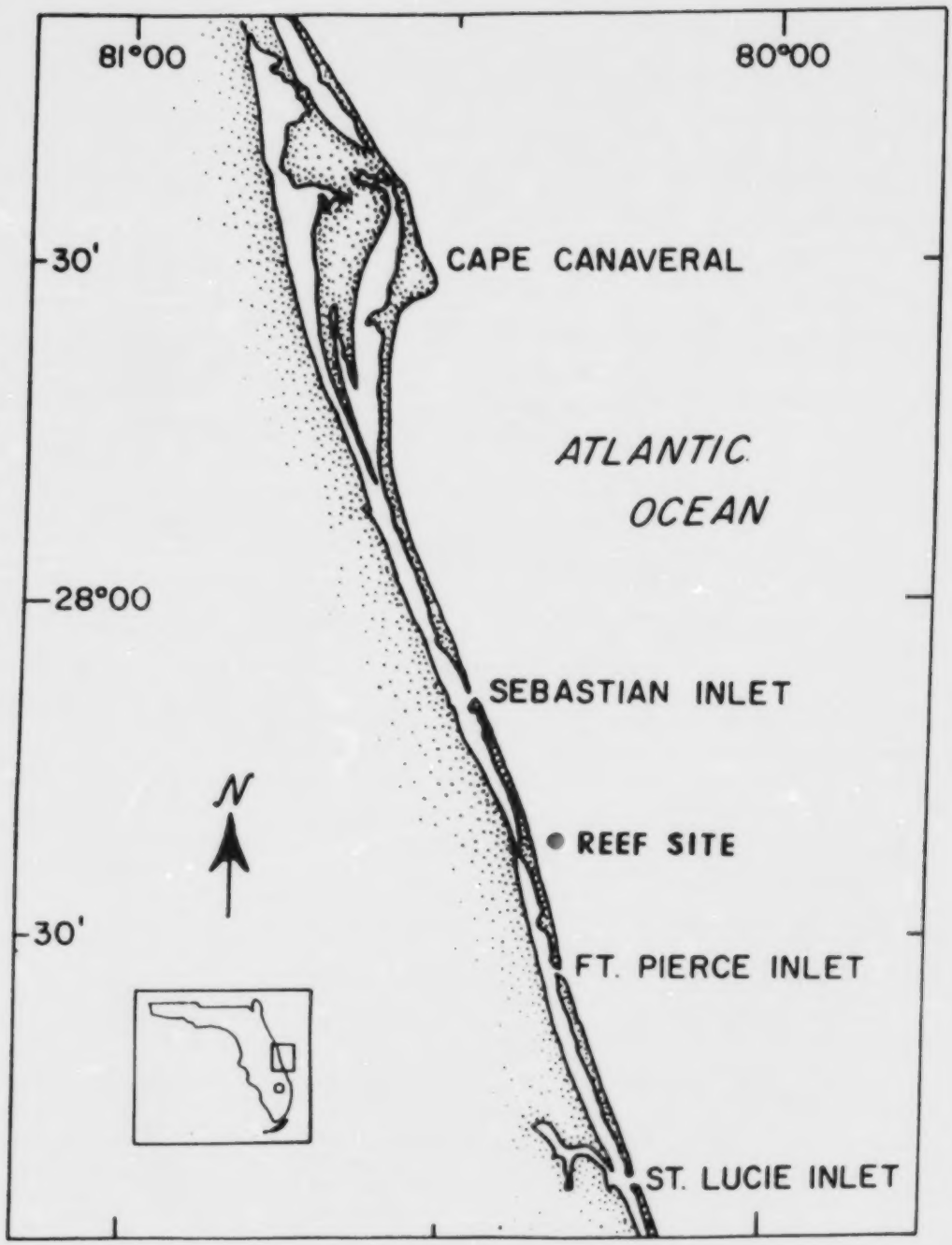

Figure 1. Experimental artificial reef location on the coast of Florida, U.S.A.

Methods

The site of the artificial reef is approximately 2 km seaward of Vero Beach, Florida, directly offshore from the Florida Institute of Technology Indian River Marine Science Research Center (Figure 1). The site is located in 12.2 m of water on a coarse sand bottom lacking in natural relief within approximately 2.5 km.

Experimental design in a study of this type is extremely difficult because of logistical considerations. However, it is inappropriate to consider a design in which only one experimental reef is constructed although this has been the case in the past (Parker et al., 1985; Woodhead and Parker, 1985). Hurlbert (1984) warns that it is impossible to get other than tentative conclusions from experiments where treatments are unreplicated. Also, taking multiple samples from a single reef is not the same as taking samples from multiple reefs of the same type when the purpose is to distinguish between specific treatment effects.

To evaluate whether the ash-blocks have any effects on the three biological communities studied, three treatments were used. These were: 1) ash block reefs, 2) concrete block reefs, 3) natural control areas without reef structures. Each treatment was replicated twice. The concrete reefs served as a control for the ash block reefs while the natural areas served as controls for the effects of the reef structures. These treatments were laid out in a 2 x 3 grid pattern to form a randomized complete blocks experimental design. A schematic of the experimental design is given in Figure 2.

Sampling for benthos and fouling organisms was carried out on a monthly basis, with fish censuses being carried out on an approximately biweekly schedule whenever visibility permitted.

Fouling Communities.

To sample fouling community development over time on the reefs, sample bricks (10 x 10 x 16 cm) were made from both the same ash material used in the large reef blocks and from regular concrete as controls. Bricks were placed by divers on top of the appropriate reefs following emplacement of the larger structural blocks. Fouling communities were sampled by the removal of three replicate bricks, selected at random, from each reef for a total of 12 bricks on each collection. Sample bricks were transported to the surface in trays.

On board the research boat, each brick was labeled and placed in an individual plastic container with lid and preserved with a 7% formalin - seawater mixture. In the laboratory, the bricks were placed in a plastic tray and covered with water to facilitate observation of the surface. The faunal composition of bricks was determined under a boom-mounted microscope by using a random point technique (Sutherland, 1974). A glass plate on which 75 random points had been drawn by computer were placed over the brick. Per cent cover of each species on the block was estimated from the number of points which fell on top of a given species. Estimates of empty space on the bricks were also made in this manner. Dead barnacles were counted as empty space available for colonization by other organisms. Total number of species per brick were also estimated from the random point technique. Additionally, counts of barnacle abundance were made by

counting all barnacles within three 9 cm^2 quadrats on each sample brick. Mortality of barnacles was estimated by counting the number of dead barnacles present in these same quadrats.

Fishes

Fishes were recorded by a diver utilizing visual census techniques. Data were recorded on underwater slates and were transferred to data sheets at the surface. For each reef, total counts of all fish species were made using the Discrete Group Visual Census method (Greene, 1985) to the best level of identification possible given conditions of visibility. Replicate counts were made in the control areas without reef structures to provide background densities of fishes by swimming 4 replicate timed transects (5 min) on compass headings. Fish were rare in these areas and this data is not reported here.

Benthos

Benthic sampling began in August, 1986, 8 months before reef installation. Benthic samples were obtained using diver operated hand-held corers of 7.6 cm diameter fitted with 0.5 mm mesh screen in one end so that they could be inverted after a core was taken. A large rubber stopper attached to each corer was used to seal the open end. Samples were placed in a basket and returned to the surface in the corers where they were transferred to labeled jars. Three replicate samples were taken within one meter of the edge of each of the four reefs, while an additional three replicate cores were taken at a distance of 2 -4 m from each reef. Cores were selected within

these areas at random. Six replicate samples were also taken at random locations within each control bottom area. A total of 36 benthic samples per collection were made. Material collected was fixed with a 7% formalin - seawater solution for 48 hours. After fixation, samples were washed on a 0.5 mm mesh sieve and all material retained was preserved in 70% ethyl alcohol containing rose bengal stain. Specimens were identified to the lowest possible taxonomic level.

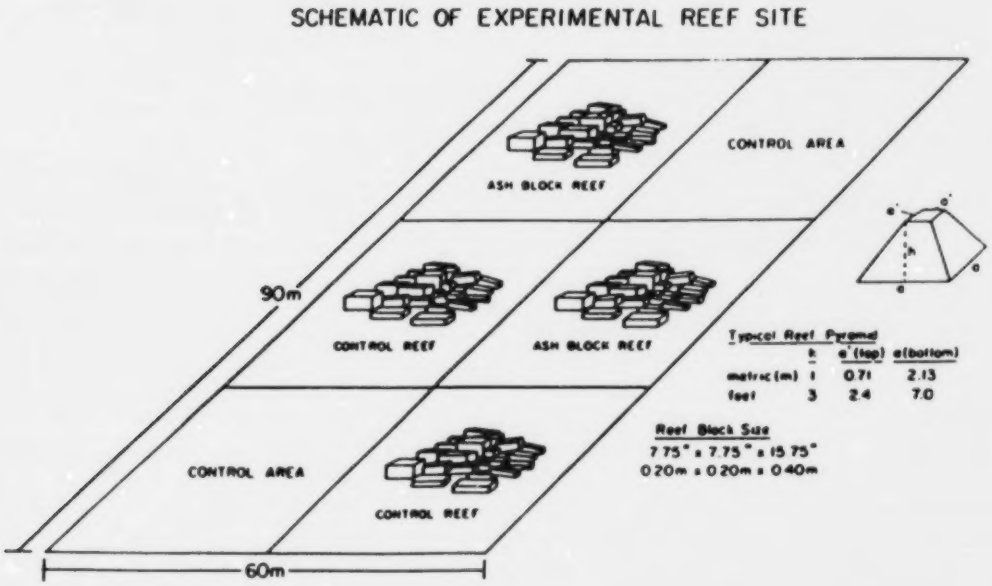

Figure 2. Schematic diagram of the experimental artificial reef array.

Results

Fouling Organisms

Settlement by fouling organisms was rapid with colonization by barnacles occurring within the first month after reef emplacement. The initial heavy settlement of juvenile barnacles resulted in there being only approximately 30% free space on the fouling bricks after one month submergence. However, free space increased with time. After four months, free space averaged 73 % on the oil-ash bricks and 68% on the concrete sample bricks (Table 1). At this time the dominant fouling organisms on both reef types were the barnacles Balanus venustus and Balanus trigonus (Table 1). There were no significant differences in per cent cover of the dominant species between the two reef types after 4 months.

Abundance of Balanus venustus decreased steadily over the first four months of reef submergence (Figure 3). There was no significant difference in abundance of this species between the two reef types on any sample date. Mortality data (Figure 4) indicated that barnacle mortality steadily increased over time.

Table 1. Comparison of the mean per cent cover of fouling organisms and empty space between oil-ash and concrete bricks taken from the the artificial reefs after 4 months submergence (July 8 1987).

	OIL ASH BRICKS	CONCRETE BRICKS
Balanus venustus	16.2	13.4
Balanus trigonus	4.5	14.9
Sabellaria sp. A	1.8	2.9
Other species	0.9	0.7
Empty space	76.7	68.2

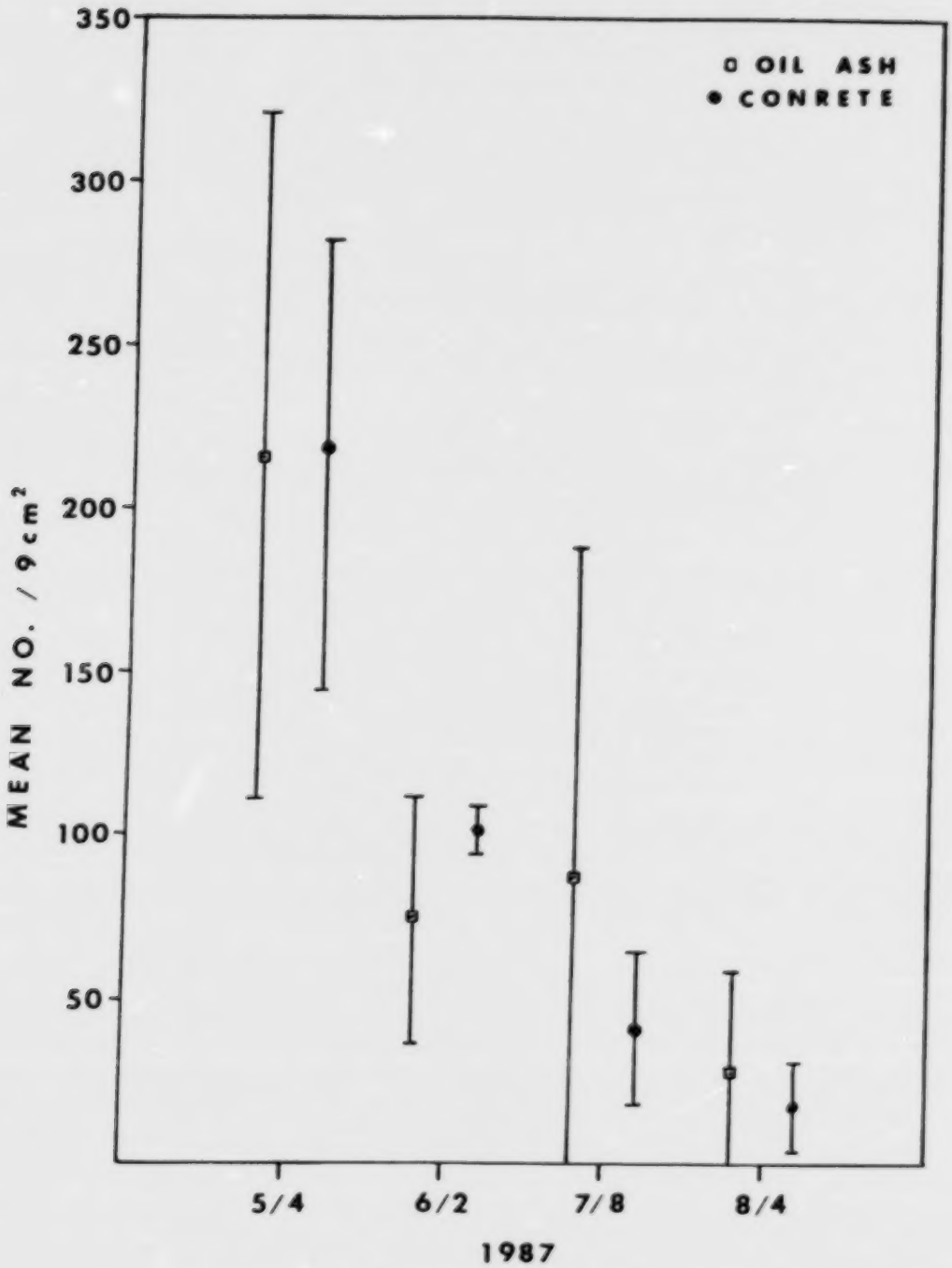

Figure 3. Mean abundance per 9 cm^2 (± 1 s.d.) of the barnacle Balanus venustus on sample bricks from the concrete and oil-ash reef treatments versus time. Data are pooled values from the two replicate reefs.

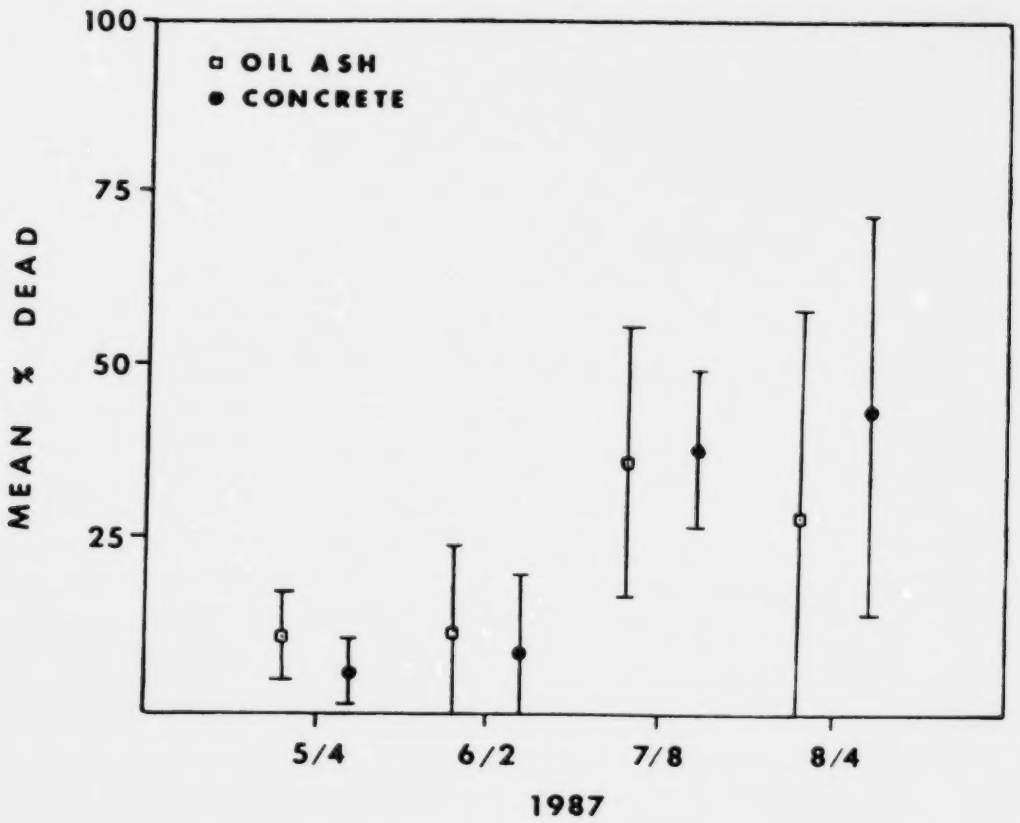

Figure 4. Mean per cent of dead Balanus venustus per 9 cm^2 (± 1 s.d.) on sample bricks from the concrete and oil-ash reef treatments versus time. Data are pooled values from the two replicate reefs.

Fish

Recruitment of fish to the reef was extremely rapid, with fish being observed on the reefs within 24 hrs. After four months a total of 34 fish species had been recorded on the reefs. Abundance was dominated by pigfish (Orthopristis chrysoptera) and by the spadefish Chaetodipterus faber.

Two weeks after reef installation, fish abundance averaged 18.5 on the oil ash reefs and 53.5 on the concrete reefs (Figure 5). By four months after installation, mean number of fish per reef had risen to 244 for the oil-ash reefs and 262.5 for the concrete reefs. There was no statistically significant difference in abundance of fish between reef type on any sample date. In June it was noted that there did appear to be some disparity in abundance between reef types. This was found to be correlated with an increased height to the reef structure for the concrete block treatments. The concrete reefs were adjusted down in height to that of the ash-block reefs at the end of June, and the difference between the two reef types greatly diminished after this date.

Mean number of species of fish per reef was 3 per reef on the oil-ash reefs and 2.5 per concrete reef after two weeks. Number of species per reef increased to 8 and 11 for the oil-ash and concrete reefs, respectively, after four months (Figure 6). There was no statistically significant difference in mean number of species of fish between reef type on any sample date.

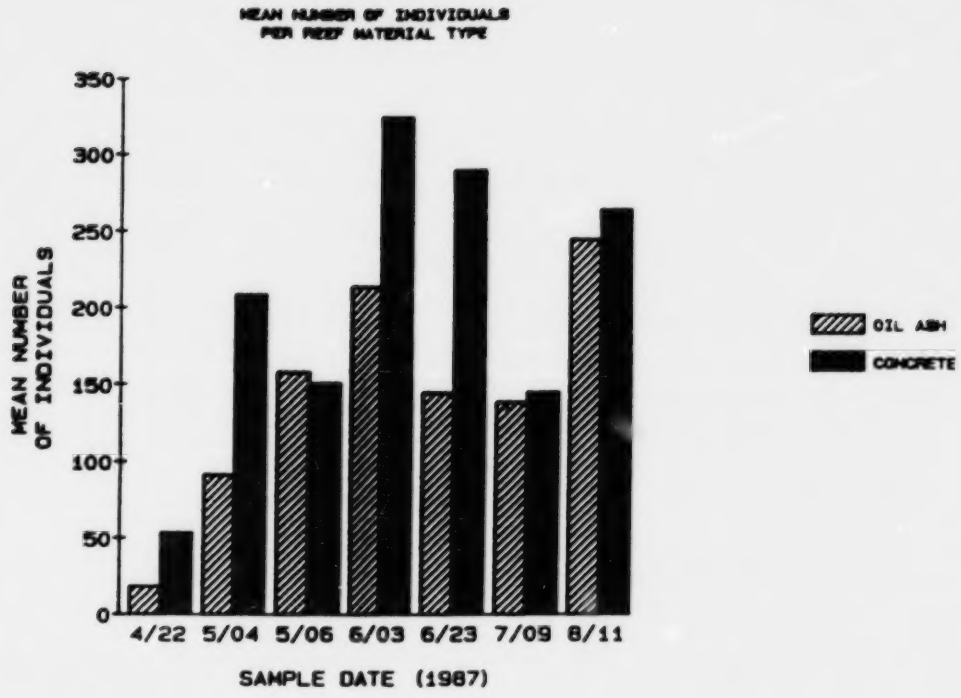

Figure 5. Mean number of fish individuals per reef from the concrete and oil-ash reef treatments versus time.

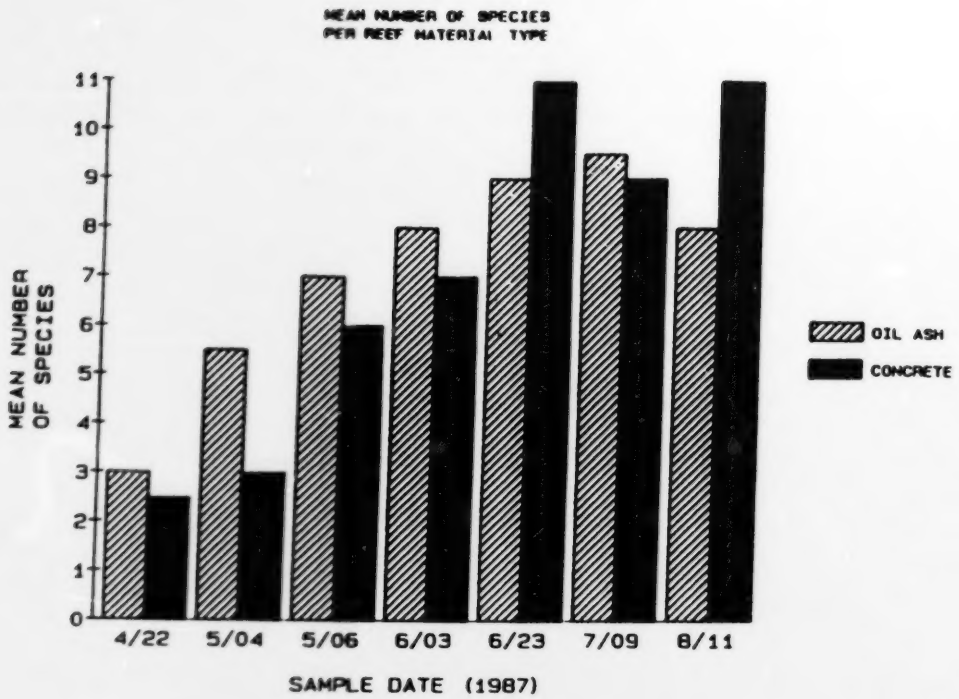

Figure 6. Mean number of fish species per reef from the concrete and oil-ash reef treatments versus time.

Benthos

At present only two months of data from the benthic sampling are available, March 22, 1987 and July 8, 1987. The first date was approximately 2 weeks before reef installation and the second was three months after reef installation. The mean number of individuals per core in the March samples was 59.8. By three months after reef installation there was an increase in average abundance of organisms in the control samples versus the mean of the pre-installation samples in March. There was little difference in abundance between the reef-associated benthic samples and the March samples. Density of benthic organisms was 57.7 in the concrete reef areas, 66.7 in the ash reef areas, and 85.3 per core in the control areas. There was a significant difference among treatments in mean abundance per core (one-way ANOVA, $F= 4.1$, $p < 0.05$), with the control areas having significantly higher abundance than the concrete reef areas. There was no significant difference in abundance of benthic organisms between the concrete and ash-block reef areas or between the ash reef and control areas (T-method of a posteriori comparisons, Sokal and Rohlf, 1981).

Comparison of the mean number of benthic taxa per core collected before and three months after reef installation showed little difference between the two collection dates. In March, there was a mean of 16.3 taxa per core. In the July sample, mean number of taxa per core was 16.7 for the concrete reef areas, 18.3 for the ash reef areas, and 19 for the control areas. There was no significant difference among the three treatments in terms of mean number of taxa for the July sample.

Abundance patterns of the benthic fauna were analysed around both of the reef treatments with respect to distance of the samples from the reefs. Mean number of individuals per core and mean number of taxa per core are compared for each reef type in Table 2. Two-way analysis of variance indicated that for both abundance and number of taxa, there were significantly fewer individuals and taxa ($p < 0.025$) for benthic samples taken within 1 m of the reef as compared to those taken at greater distance. There was no significant difference between reef types in either abundance or number of taxa. These results indicate that the presence of the reef structures, regardless of block composition, causes a relative decrease in abundance and taxa number immediately adjacent to the reefs.

Table 2. Comparison of mean number of individuals and mean number of taxa per core between reef types and between samples taken within 1 m and at greater than 1 m distances from the reefs three months after reef emplacement (8 July 1987).

SAMPLE	REEF TYPE	
	ASH	CONCRETE
Mean No. Ind./Core <1m	50.3	54.0
Mean No. Ind./Core >1m	83.0	61.0
Mean No. Taxa/Core <1m	14.5	15.0
Mean No. Taxa/Core >1m	21.0	18.0

Discussion

After four months of submergence, there was no indication of any significant difference between the fouling community which had become established on the oil-ash as compared with the concrete block reefs. This was the case for per cent cover of organisms, and for density and mortality of the dominant barnacle species. The decrease in abundance per unit area over 4 months observed for Balanus venustus might have been due to growth of individual barnacles which would result in a decrease in absolute numbers while per cent cover remained high. However, per cent cover of Balanus venustus also decreased steadily after two months of submergence. The mortality data for B. venustus confirmed that it was the loss of large numbers of barnacles and not growth of barnacles which decreased barnacle density.

Fish abundance and species richness showed patterns of increase over time which were similar for both reef types, with no significant difference being found between reef types on any sample date over a four month period. Benthic organisms, although examined only after three months following reef installation, also showed no significant differences between the reef types. It appears that the biological communities respond to the ash block reefs in the same fashion as for the concrete block reefs. There was no evidence from any of the three biological communities analysed of detrimental effects of the oil-ash blocks after the time period examined.

Biological interactions do appear to be significant in determining the structure of the communities associated with the

reefs. The increase in fish abundance over time on the reefs appears to have resulted in increased predation pressure on the fouling community on the reef blocks, in turn resulting in a decrease in per cent cover of organisms in the short term. Sea urchins which recruited to the reefs and might have affected the fouling community disappeared within a few months, presumably because of the recruitment of such fishes as triggerfishes to the reefs. Whether per cent cover of organisms will increase as organisms such as sponges and colonial tunicates, which may be less palatable to fish, recruit to the reefs remains to be seen.

Along with the decrease in the fouling community, the benthic infaunal community in the area immediately adjacent to the reefs was less abundant than that located further away. While a physical cause can not be ruled out at this point, a possible explanation of this effect may be the foraging of fishes and benthic invertebrates attracted to the reefs. Portunid crabs were observed recruiting to the reefs within two days of installation, and together with benthic feeding fishes such as the pigfish which had recruited in large numbers after one month, may have resulted in an increase in predation pressure with a resultant decrease in benthic organisms. Such a halo pattern is consistent with predation pressure which originates from benthic feeding fish which use the reef structures for shelter and forage out from it. Recent information from a low relief rock structure associated with a sewer outfall in Monteray Bay, California also indicated that significant changes in the infaunal community occured only within one meter of the rock structure (J. Oakden, Moss Landing Marine Laboratory, personal communication), although the cause

of this response was not clear.

Recent recruitment of large predators such as barracuda to the artifical reefs may result in a decrease of other fishes on the reefs. This in turn may result in changes in the fouling and benthic communities in the longer term.

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POLYCYCLIC AROMATIC HYDROCARBONS ASSOCIATED WITH AN ARTIFICIAL REEF CONSTRUCTED OF COAL AND OIL ASH WASTE PRODUCTS.

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Abstract

Polycyclic aromatic hydrocarbons (PAH) occur widely in the lithoshere and have been reported in various soils, marine sediments, and in urban limnic sediments in the United States and Canada. They have been found to be associated with fly ash emitted from and retained by coal fired electric power plants. As the temperature decreases both in the stack and as the fly ash leaves the stack, the PAH readily adsorb onto the surface of the fly ash and may become available to the environment. Waste material from coal fired power plants disposed of in the marine environment is of concern for a number of its chemical components, including PAH. A similar scenario could be drawn for oil ash waste. Specific PAH which have been identified on residues of fossil fuel combustion include anthracene and phenanthrene, fluoranthene and pyrene, chrysene and benzanthracene, benzofluoranthenes and benzopyrenes. Each of these pairs of isomers has differing chemical and physical properties which make generalization about all PAH difficult. Stabilized coal and oil ash wastes have been used to construct small-scale artificial reefs in the Atlantic Ocean off Indian River County, Florida. These reefs, composed of waste and reference materials, are the subject of an intensive environmental impact study. Components used to construct building blocks for the reef were individually analyzed for their PAH content. The results of the GC analyses of the ash suggest relatively high concentrations of PAH derived from both combusted and uncombusted fossil fuel. The distribution of the PAH indicates that low molecular weight PAH are also present at relatively high concentrations. Analysis of formed reef construction blocks show similar patterns for the PAH. However, the reef blocks yielded lower concentrations of PAH upon extraction. Concerns for the relative solubility and toxicity of these low molecular weight PAH and other organic constituents associated with the oil and fly ash material resulted in recommendations to study the leaching behavior of representative PAH from the stabilized ash matrix in typical marine environments.

Introduction

Charles I banned the burning of coal within the city limits of London in the 13th century during periods of

excess fog. The penalty for the violation of this edict was public beheading. This is the first record of a serious health impact associated with the combustion of fossil fuel. Polycyclic aromatic hydrocarbons (PAH) are a series of aromatic hydrocarbons which consist of condensed ring systems. They are generally composed of two or more benzene rings fused in linear, angular or cluster arrays such that adjacent rings have either 2 or 3 carbons atoms in common. Five carbon atom rings may also be included in these structures (Blumer 1976). A typical PAH is naphthalene which consists of two fused benzene rings. As the number of condensed rings increases, the number of possible isomers also increases. Alkylation of the rings will also greatly increase the number of possible isomers.

Polycyclic aromatic hydrocarbons have been found extensively in the marine, freshwater and sedimentary environment. Concentrations of PAH from urbanized areas may be greater than 100,000 ppb while rural soils and ocean sediment samples far from the influence of man have levels of PAH less than 100 ppb. (Hites et al., 1980 a, b; Laflamme and Hites 1978; Windsor and Hites, 1978; Giger and Blumer, 1974; Giger and Shaffner, 1978, Wakeham et al., 1980). Polycyclic aromatic hydrocarbons may be generated during the incomplete combustion of wood or fossil fuels such as coal or oil (Natusch, 1978). At coal fired electric power plants, PAH have been found to be retained in the fly ash. As the highly sorbtive fly ash is emitted from the plant stack, the PAH is readily available to the environment and can be transported great distances. Since the unsubstituted homologs are formed preferentially at high temperatures, they will be dominant in the ash (Blumer, 1976).

Polyclic aromatic hydrocarbons in the environment are of interest because of their potential toxic nature. The larger molecular weight compounds, such as benz(a)pyrene, are known to be carcinogenic in laboratory test mammals (Searle, 1976). The smaller ring compounds, although not known to be carcinogenic, exhibit acute toxicity to marine organisms (Anderson et al., 1974). PAH has been found to be taken up by marine organisms and can accumulate in their tissues (Obana et al, 1981; Larsen et al., 1983).

The efficacy of disposing wastes containing toxic substances of environmental concern such as PAH is limited by how effectively the toxic substance is removed from the biological cycle. Various methods have been used for disposal of coal and oil power plant wastes with most centering on landfill options. Direct dumping of the power plant wastes into the ocean can result in high turbidities which can directly disrupt benthic communities. The waste ash may also contain toxic substances such as trace metals which can potentially leach out of the ash into the seawater (Duedall et al., 1985; Breslin, 1986).

A novel method for disposal of coal ash has been its consolidation with hydrated lime and cement in the form of hard blocks. The formation of the blocks results in the

chemical stabilization of the ash and has enabled their use for artificial reef construction in the marine environment. Studies have indicated that, after 80 weeks of immersion in seawater, the blocks keep their physical integrity and trace metal concentrations on the blocks were found not to have significantly changed (Parker et al., 1984; Parker et al., 1985). Species richness on the stabilized blocks was not significantly different from control blocks and no increase was found in uptake of toxic metals by mussels mytilus edulis (Humphries et. al., 1985).

A small scale artificial reef has been constructed in the Atlantic Ocean off Indian River County in Florida using the stabilized coal and oil ash and is the subject of an intensive environmental study. Although the chemical composition and the trace metal leaching characteristics for numerous varieties of coal ash and stabilized blocks are fairly well known, few data exist on the amount of leaching of any organic constituents including PAH both from the ash and the stabilized blocks. The ability of seawater to leach PAH contained in coal ash is directly related to the environmental feasibility of using the stabilized ash for disposal in artificial reefs. The main purpose of this study will be to determine the amount of PAH present in the ash and stabilized block and therefore the amount which can potentially be leached from the ash and the block by seawater.

Experimental

Samples of fly ash and oil ash were obtained from Florida Power and Light and analyzed to determine specific PAH present and their concentration levels. The oil ash was a brown wet solid with large clumps of material while the fly ash was a wet gritty solid with no clumping of the material. Both samples were analyzed as received. The extraction and analysis procedure used is reviewed by Laflammme and Hites (1978) and Gschwend and Hites (1981) and is based upon a nitromethane partition described by Hoffman and Wynder (1960). Samples of the fly ash and oil ash (100-200 grams) were soxhlet-extracted with toluene-methanol (1:1) for 48 hours with fresh solvent added after 24 hours. The toluene and methanol were removed under vacuum and the remaining water was extracted with five volumes of hexane. The hexane extracts were combined and evaporated under vacuum. The sample was then redissolved in a mixture of cyclohexane-nitromethane (1:1) and extracted with five volumes of nitromethane. The nitromethane extracts were combined and removed under vacuum. The extract is then purified by column chromatography on two grams of silica gel and elution with 300ml of hexane. The hexane is removed and the eluate is dissolved in methylene chloride for subsequent analysis for PAH by gas chromatography.

Stabilized ash blocks were obtained to determine if comparable levels of PAH exist in the blocks. The stabilized ash blocks were slightly larger than typical bricks used in home construction. They consisted of approximately 35% each of fly ash and oil ash with the remaining 30% being the concrete matrix. As a result of the large ash content, the blocks are soft in composition. Control blocks of the same size made entirely of concrete were also analyzed for PAH concentrations. The surface of each block was mechanically cleaned to remove any atmospheric dust accumulation which may contain PAH. Whole blocks were then ground and extracted by the identical procedure for final analysis by gas chromatography. Procedural blanks were checked for each set of sample extractions.

Gas chromatography was conducted using a Sigma 3B Gas Chromatograph with a flame ionization detector. The column was a fused silica capillary column SE-54 with a nitrogen carrier gas. The temperature program ran from 85 degrees C to 275 degrees C at a rate of 10 degrees C/min. Individual PAH compounds were identified by relative retention times using standards for conformation. Quantification was made by measurement of peak areas and comparison to standard calibration curves. Deuterated anthracene was used as an internal standard to determine the percentage of recovery. All concentrations have been corrected for blanks and internal standard recovery which ranged from 51 to 88%.

Results and Discussion

From the initial sample analysis, six specific PAH were chosen for routine analysis: naphthalene, fluoranthene, pyrene, chrysene, benzo(a)pyrene and perylene. Based upon standard retention times, no PAH have been identified as being present in the control blocks above blank levels. The concentrations found in the oil ash, fly ash and the stabilized blocks are reported in the table. The oil ash contained the highest levels of PAH with concentrations ranging from a high of 1200 ppb for fluoranthene to 270 ppb for naphthalene. Concentrations in the stabilized ash blocks were considerably lower with only approximately 250 ppb of fluoranthene and 50 ppb of naphthalene. Levels of PAH in the fly ash were intermediate to those concentrations found in the oil ash and the stabilized blocks. Perylene and benz(a)pyrene were identified as being present in both the oil ash and the fly ash while neither compound was detected during analysis of the stabilized block.

The concentrations of PAH found in the ash samples and the blocks are relatively high compared to levels of PAH found for various world sediments. The high levels of PAH in the ash samples are comparable to samples from areas which have a high urban influence such as New York Bight which has a concentration of PAH of 5800 ppb for the same constituents (Hites, Laflamme and Windsor 1980). The

relative concentrations of naphthalene, fluoranthene, pyrene and chrysene are consistent from the ash samples and the block samples analyzed. However, since the stabilized blocks are composed of approximately 35% each of oil ash and fly ash and 30% concrete, it should be noted that the results for the concentrations of PAH in the stabilized blocks are lower than expected. This appears to be due to the concrete matrix effectively containing the PAH and reducing the amount removed during the extensive extraction process. A key difference from PAH concentrations in worldwide sediment samples is the presence of napthalene in the ash and block samples. This may be due to incomplete combustion of the fuel or post generation contamination of the ash at the waste holding sites.

Because of the great diversity of PAH and the number of isomers, their chemical and physical properties can be highly variable. This is reflected in the aqueous solubility of specific PAH. In general, the solubility in fresh water decreases as the number of condensed rings on a species increases. For example, at 25 degrees C, naphthalene has a solubility of approximately 32mg/kg while the 5 ring benz(a)pyrene has a solubility of only 0.009mg/kg. Alkylation also tends to decrease the solubility of a particular PAH while increasing temperature increases the solubility of a particular PAH (May 1980). Since the presence of relatively high concentrations of highly soluble naphthalene and other low molecular weight PAH have been found in our samples, there is concern for the leaching of the PAH in seawater. Some variation between seasons in the natural environment or between tropical, temperate and polar regions may be expected for this or for similar reefs.

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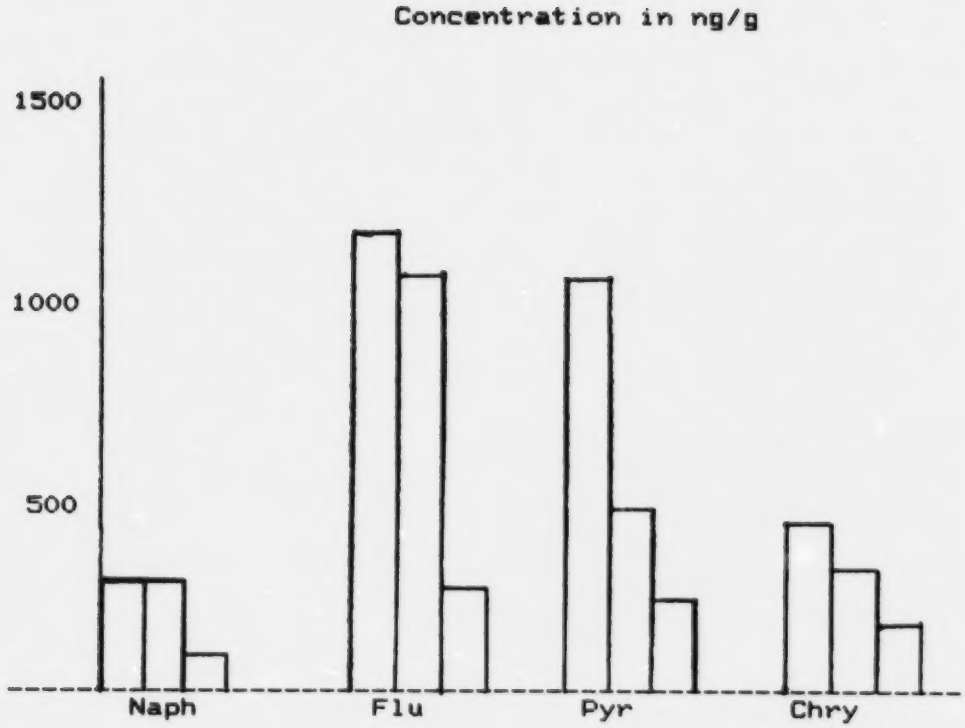
TABLE

Concentration in ng/g

Sample	Naph	Flu	Pyr	Chry	BaP	Pery
Oil Ash	270	1200	1100	480	420	310
Fly Ash	270	660	490	300	320	160
Block A	48	240	210	200	ND	ND
Block B	50	260	190	120	ND	ND

ND = not detected above blank levels
All Standard Deviations are less than 10%

Naph = Naphthalene
Flu = Fluorene
Pyr = Pyrene
Chry = Chrysene
Bap = Benz(a)pyrene
Pery = Perylene



Naph = Naphthalene
Flu = Fluoranthene
Pyr = Pyrene
Chry = Chrysene

Oil Ash - Left
Fly Ash - Center
Block - Right

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ISSUES AND CONCERNS SURROUNDING THE OCEAN DISPOSAL OF FISH PROCESSING WASTES AND A CASE STUDY

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A significant proportion of processing waste from the Atlantic Canadian fishing industry has traditionally been disposed of in the ocean. While a longstanding debate still exists regarding the advisability and best methods for ocean disposal of this material, other options for fish offal disposal are diminishing.

In 1986, herring carcasses were disposed of in Southwestern Nova Scotia through ocean dumping, landfill, agricultural applications, fish meal and composting. Difficulties with all but the ocean dumping and fish meal options resulted in these options being forbidden by local municipal governments for the 1987 season. Fish meal plant capacity is still inadequate to handle the majority of waste from the area and therefore the requirements for ocean dispossal are increasing.

This paper discusses the issues and concerns surrounding the ocean disposal of fish processing wastes in Eastern Canada. Also, application of the decision-making process employed in creating guidelines for the disposal of herring carcasses resulting from a roe herring fishery in southwestern Nova Scotia is presented.

In an effort to design the most acceptable ocean dumping programms for the regions, consultations were held with various government agencies, processors, community groups and fishing industry representatives. The primary issues in this case were fouling of lobster grounds, baiting of lobster grounds, fouling of herring grounds and gear, and the possibility of herring carcasses being washed ashore.

A discussion of previous fish offal research projects in Newfoundland and New Brunswick is presented. Preliminary results of direct observation experiments to be conducted during the disposal of wastes from the 1987 herring roe fishery are discussed.

Disposal of fish waste at sea has I suppose been going on as long as man has fished from boats. As the intensity of fishing has increased so has fish harvesting, and the need for bulk processing has resulted.

Uses of fish waste have developed over the years from such simple uses as fertilizer and bait to more sophisticated uses in fish silage, fish meal and compounded feed for livestock and fish farming. However, where transportation of raw material is difficult, processing facilities unavailable or if economic disincentives such as low fish meal prices are prevalent, disposal of the waste then appears to be the only viable alternative. Dumping over the end of wharves, whilst practical for very small quantities of waste, quickly leads to smelly anoxic conditions resulting in fish being out of sight but definitely not out of mind! Landfill in municipal dumpsites or dumping in abandoned quarries or gullies has been employed, but transporting large volumes of material and the associated odours of exposed piles and attracted pests have discouraged this practice to the extent that many communities which once used this method have now abandoned it.

Because of the thin soil and because deep waters are within easy access of the shore, "gurry" disposal at sea has long been favoured in Newfoundland. In 1983, the Regional Ocean Dumping Advisory Committee, which advises on ocean dumping permit issuance, sponsored a study to determine the prevalence and extent of fish waste disposal in Newfoundland. The results are summarized in the following table.

1982 QUANTITIES OF OFFAL

1982 QUANTITIES OF OFFAL DISPOSED BY OCEAN DUMPING

KEY: Status - C - Current
H - Historic

LOCATION	OPERATOR	STATUS	QUANTITY IN LBS.
AREA 1			
Barenced	Argosy Seafood Limited	C	2,190,749
Bonavista	(Crab) The Lake Group Ltd.	C	1,400,000
Chance Cove	Smith Seafoods Limited	C	?
Coley's Point	Coley's Point Fisheries Ltd.	C	?
Coley's Point	Seaside Fisheries Limited	C	750,000
Dover	Ichthus Sea Products	C	?
Gooseberry Cove	Gooseberry Cove Fisheries	C	?
Greenspond	Greenspond Fish Processors Limited	C	1,580,000
Hant's Harbour	P. Jones & Sons Limited	C	500,000
Happy Adventure	Happy Adventure Sea Products Limited	C	500,000
Port de Grave	Ocean Harvesters Limited	C	2,932,206
Quidi Vidi	Cabot Seafoods Limited	C	750,000
New Harbour	(Caplin) Higdon's Seafoods	C	?
Trouty	Bonaventure Seafoods Ltd.	C	50,000
Valleyfield	(only excess) Exothic Fish Processors Limited	C	2,640,000
AREA 2			
Argentia	Blue Ocean Products Limited	H	?
Bellevram	Fishery Products Limited	H	?
Gaultois	The Lake Group Limited	H	?
Hermitage	Fishery Products Limited	H	?
Jerseyside	(Indiv. fishermen) Placentia Bay Fisheries	H	?
Lawn	H.E. Dawe Limited	C	?
Placentia	Placentia Dock	H	?
Ramea (2 sites)	The Lake Group Limited	H	?
St. Bride's	Cape Pine Fisheries	C	400,000
Southern Harbour	Port Enterprises Limited	C	2,500,000
AREA 3			
Side Arm	The Lake Group Limited	C	725,000
Bridgeport	North Atlantic Fisheries Ltd.	C	?
Comfort Cove	Norte Dame Bay Fisheries Ltd.	C	?
Conche			

This study was followed in 1985 by a closer look at the practices of three of these plants in particular. The results of this study, which was carried out by L. G. L. Ltd., indicated that although use of deep water sites were probably an acceptable disposal method, disposal at an in-harbour dumpsite in Fogo Island Harbour did result in the accumulation of fish wastes on the sea floor, where they remained for an extended period of time, even years and were not readily assimilated by the ecosystem. In other words, part of the material dumped at the dumpsite remained undegraded and accumulated, resulting in anoxic conditions and wastes which are unuseable by scavenge fish and crustacea.

In Nova Scotia and New Brunswick in 1986 and again in 1987, there have been large amounts of waste herring in the order of 50 to 100 thousand metric tons, bi products of the lucrative fall herring roe fishery which derives from the growing demand by Japanese markets for the East coast roe products (West Coast supplies being already severely over taxed).

In this fishery, all male fish and everything except the roe (the full ripe ovaries) of the female herring are of secondary value. The roe of the female herring commands market prices considerably greater that the entire former value of the herring fishery when it supported a whole-fish product industry or meal production.

Although uses for some of this waste fish have been found, the supply, during the short harvest season of two to four weeks, greatly

exceeds the demand or at least the capability of productively handling it.

The growing unacceptability of land disposal and the absence of suitable or adequate processing capacity for the waste fish has forced plant managers to consider marine disposal as an alternative.

A series of meetings was held at the request of the municipalities concerned and with representative of various fishing interests to consider the alternatives. In 1986 landfill was extensively employed, but problems associated with this practice led all but one community to abandon this approach. Many citizens had lived too long with the odour of fish reduction and meal production and were skeptical that odourless operations could be achieved.

The herring fishermen themselves believed that dead herring in the water would provoke an avoidance reaction in live herring and discourage them from spawning in the area. They were also anxious that dead herring not be dumped on the spawning grounds. No information was found to support or refute their concerns and this could be a useful area of reseach.

Lobster fishermen were mixed in their reaction, some wanted dumping, or at least the spreading of herring waste in their area, to feed lobsters and encourage them to remain in the area for an extended time, until the herring season was over. Then as waste herring would no longer be available, the lobsters would be drawn to the lobster trap bait which

could by then be deployed. Again this interesting idea was not supported by experimentation.

Other lobster fishermen were against the dumping of herring on lobster grounds on the basis that it would give an unfair advantage to those fishing in the areas where dumping took place.

Yet others felt that the lobsters would be negatively affected in some way by the dumping. On the basis of these concerns, a protocol was drawn up for herring disposal. This protocol is found in Appendix 1.

MONITORING IN 1987

After much consideration, it was decided that only a dumpsite 20 nm offshore would be approved for use in 1987 since there was too much public opposition to a near-shore dumpsite. However, an experimental dumping of 40 tons of herring waste was carried out 10 nm off the coast in 61 m of water and dispersed in the manner prescribed in the protocol. The results were video recorded using an underwater remote controlled camera carried on a "Dart" remote controlled vehicle which was owned and operated by the Department of National Defense (DND) who assisted us on this occasion. The results of this experiment are currently being written up by Martec Ltd., however, for those that are interested selections of the video footage are being shown during the mud video session.

The evidence of this research is that the dispersed herring rapidly sank (at a rate of approximately 10 cms/sec). By the time the material

reached the sea floor, it was dispersed to a density of about one carcass per five square meters in the immediate dumpsite area, and bottom currents of about 10 cm/sec were further dispersing the material.

Almost as soon as dumping occurred dogfish sharks (Squalus sp.) appeared in the area in rapidly increasing numbers. Active feeding was observed. A visit to the dumpsite the following day showed a total absence of herring waste and a large number of expectant dogfish.

These results were encouraging in that they indicate that:

1. Rapid assimilation of waste herring by the ecosystem is probable, especially by dogfish.

2. The likelihood of waste accumulating on the sea floor is slight when the protocol is observed.

3. Little or no waste remains for lobster scavenging.

4. Little or no herring waste remains to affect herring spawning behavior.

5. The possibility of a harvest of dogfish exists.

In 1988, we are optimistic that progress will have been made on finding useful alternatives for herring carcasses which will change waste to a marketable commodity. Nevertheless should the need for ocean disposal again arise, we are now in a good position to recommend an economical disposal methods.

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APPENDIX 1

GUIDELINES FOR THE DISPOSAL OF HERRING CARCASSES AT SEA SOUTHWEST NOVA SCOTIA, 1987

1) The ocean dumping of herring wastes should only be considered if there are no alternative uses for the herring, such as food, meal, bait, fertilizer or silage.

2) No dumping of herring carcasses at sea will be permitted without a valid Ocean Dumping Permit.

3) To obtain an Ocean Dumping Permit, the applicant should contact the Marine Disposal Section of the Environmental Protection Service, Conservation and Protection, Environment Canada (see Sec. 20 for the complete address) for the appropriate application form. The completed application form must be submitted to the same address together with four copies of a map or chart showing the exact location of the proposed loading site and dump site(s). In addition, a fee of $50.00, payable to the Receiver General of Canada, is required for each application.

4) Applications should be submitted well in advance of the proposed dumping operations. At least four to six weeks must be allowed for the processing of the application. This processing includes a comprehensive technical review within different Federal and Provincial agencies, and the official publishing of the Permit in the Canada Gazette.

5) Ocean Dumping Permits for the disposal of herring wastes will be issued only for the current herring fishing season and will be valid only for the period designated on the Permit unless an amendment is approved.

6) Permits for the disposal of herring carcasses cannot be used for the disposal of other fish wastes unless otherwise specified in the Permit or approved by the Regional Director of the Environmental Protection Service, Conservation and Protection, Environment Canada, Dartmouth.

7) The Permit is non-transferable. It is only valid for the Permittee or his designated carrier as indicated in Condition 1 of the Permit.

8) The Permit is only valid for total quantities up to the maximum indicated in Condition 10 of the Permit. If this quantity is likely to be exceeded, the Permittee must contact Environment Canada to apply for an amendment for the Permit. This procedure holds true for any changes in Permit conditions. Again, reasonable time should be allowed for the processing of a Permit amendment.

9) An offshore dump site 25 miles off Yarmouth, has been designated for the disposal of herring wastes and was used in 1985 and 1986. The site is delineated by a triangle, the corners of which are located by the following coordinates:

1. 43°43.00'N; 66°43.00'W;
2. 43°48.50'N; 66°53.30'W;
3. 43°45.50'N; 66°55.40'W.

It is also delineated by Loran C lines:

1. 9960-W-12220;
2. 5930-X-13220;
3. 5930-Y-31360.

The site is considered by both Environment Canada and the Department of Fisheries and Oceans as the most appropriate site for large quantities of herring offal dumping.

10) Following evaluation for potential impacts on fish stocks, spawning grounds and lobster grounds, one or more experimental nearshore dump sites may be considered for smaller (less than 500 metric tonnes) dumping operations in 1987. These dump sites will be the subject of a monitoring and evaluation program and will be re-assessed at the end of the season. Results will be made available before the commencement of the 1988 herring season.

11) Dumping in anticipation of being issued a Permit, while the application is being reviewed, will not be tolerated. Violations may result in fines (see the attached Ocean Dumping Control Act and Regulations for the total amount of fines for illegal dumping), and in the application being rejected.

12) The location, quantity and timing of each instance of dumping must be reported to the Yarmouth office of the Department of Fisheries and Oceans prior to the vessel leaving for the dump site. This reporting provision will be useful in determining how much herring was actually disposed of at different sites. It will also help in monitoring the herring disposal operation.

13) Herring should be dumped within 24 hours of processing for roe removal to minimize bloating and thus the probability of carcasses floating or reaching shore. If this is not possible, the herring must be stored in a manner which would prevent or reduce bloating and decomposition.

14) The wharf shall be kept clean and loading should be carried out in such a manner that fish carcasses do not spill onto the wharf or into the harbour waters. Failing to comply with this requirement could result in fines or suspension of Permits.

15) Routine inspections are needed to monitor environmental effects and to ensure that conditions of the Permits are being met. The Permittee will therefore, when requested, allow an Ocean Dumping Inspector to come aboard for monitoring a dumping operation.

16) Herring can be disposed of by using a pump or by using a bottom-dump barge. Whole herring may only be dumped at the offshore dump sites. Nearshore sites, if approved, may only be used if the herring carcasses are macerated, that is, chopped into pieces.

17) Herring should be dumped or pumped from the carrier in a controlled manner while the vessel is steaming through the dump site at maximum safe speed. If a barge has more than one pocket, dumping should be carried out one pocket at a time, with a pause of several minutes between pockets, while the tow boat is steaming. This will promote dispersion and will not lead to clumps of decaying fish on the seafloor which might remain for a period of time and adversely affect the seafloor and associated organisms.

18) Within thirty days after the final dumping, the Permittee must report in writing to the Regional Director of the Environmental Protection Service, Conservation and Protection, Environment Canada, Dartmouth. This report must record the date, location and quantity for each dumping incident carried out under the Permit.

19) Permitting of herring carcass disposal will be reviewed on an annual basis until a satisfactory alternative is found.

20) For further information, contact Alan McIver or David Aggett at (902) 426-8301, or at the following address:

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(May 1987)

Phaseout and closure of the 12-mile sewage sludge dump site: a case study on ecosystem response

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ABSTRACT

A major issue facing policy makers charged with resolving problems of ocean disposal of sewage sludge is the uncertainty as to whether closure of existing nearshore sites will eventually result in ecosystem changes significant enough to justify designation of more costly offshore locations. To date, the only existing information on such changes comes from surveys conducted prior to and following closure of the highly dispersive Philadelphia site which indicated that after four years, microbial contamination had returned to background levels. The scheduled closure of the depositional 12-mile sewage sludge dump site in the New York Bight thus offers a unique opportunity to document changes in the habitat and biota during and following planned phaseout and cessation of dumping by December, 1987. With collaborative support from several federal, state and university groups, surveys of 25 stations in and around the dump site were begun by investigators of the Northeast Fisheries Center/NMFS/NOAA in July 1986. Choice of variables was based on their relevance to fisheries in the inner New York Bight as well as predictive value for detecting change as dumping is abated. Physical, chemical and biological approaches have been developed and variables to be measured include water and sediment chemistry, benthic community structure, distribution, abundance, gross pathology and food habits of fishes and invertebrates, body burdens of selected contaminants, microbial contamination of shellfish and sediment as well as pathways and rates of dispersion of sludge and contaminants. A number of working hypotheses have been developed by which recovery will be assessed, e.g., levels of shellfish bacteria indicative of recent sewage contamination will decrease to acceptable levels permitting opening of closed beds; diets of selected species, including lobster and winter flounder will change as a direct consequence of shifts in benthic forage species; the distribution and density of Capitella, a polychaete indicative of organic enrichment will be reduced at certain stations. Preliminary findings from the first year's surveys are presented.

INTRODUCTION

Successful management of coastal resources is often hindered by problems arising from conflicting multiple use. Nowhere is this more evident than in the New York Bight (Mayer, 1982; Squires, 1983). In the wake of the numerous tankers and freighters which sail into the harbor each day, commercial and recreational fishermen compete for limited stocks of fish and shellfish, while during the summer, many of the area's 18 million residents crowd the beaches in search of clean sand and water. Access to and utilization of these and other natural resources are compromised, however, by other uses of the Bight - particularly as a relatively inexpensive disposal site for the dredge spoils, sewage sludge and other wastes from New York and New Jersey (Fig. 1). Resolution of these conflicts involves not only questions of cost but also of public health.

The issue of public health arose in May 1970, when the U. S. Food and Drug Administration (FDA), acting under the National Shellfish Sanitation Program, closed an area within an 11-km radius of the 12-mile sewage sludge dump site to commercial shellfish harvesting (Fig. 2; Verber, 1976). This followed findings of elevated levels of coliform bacteria, indicating sewage contamination and possible presence of pathogenic organisms. In addition, a determination that municipal sewage sludge dumped at this site was a major source of sewage-related contaminants in the Christiaensen Basin and Hudson Shelf Valley, prompted the U.S. Environmental Protection Agency (EPA), under an amendment to the Marine Protection, Research and Sanctuaries Act (MPRSA) to deny applications for further dumping at the site after December 31, 1981 (for review see Erdheim, 1985; Santoro, 1987). The site, in

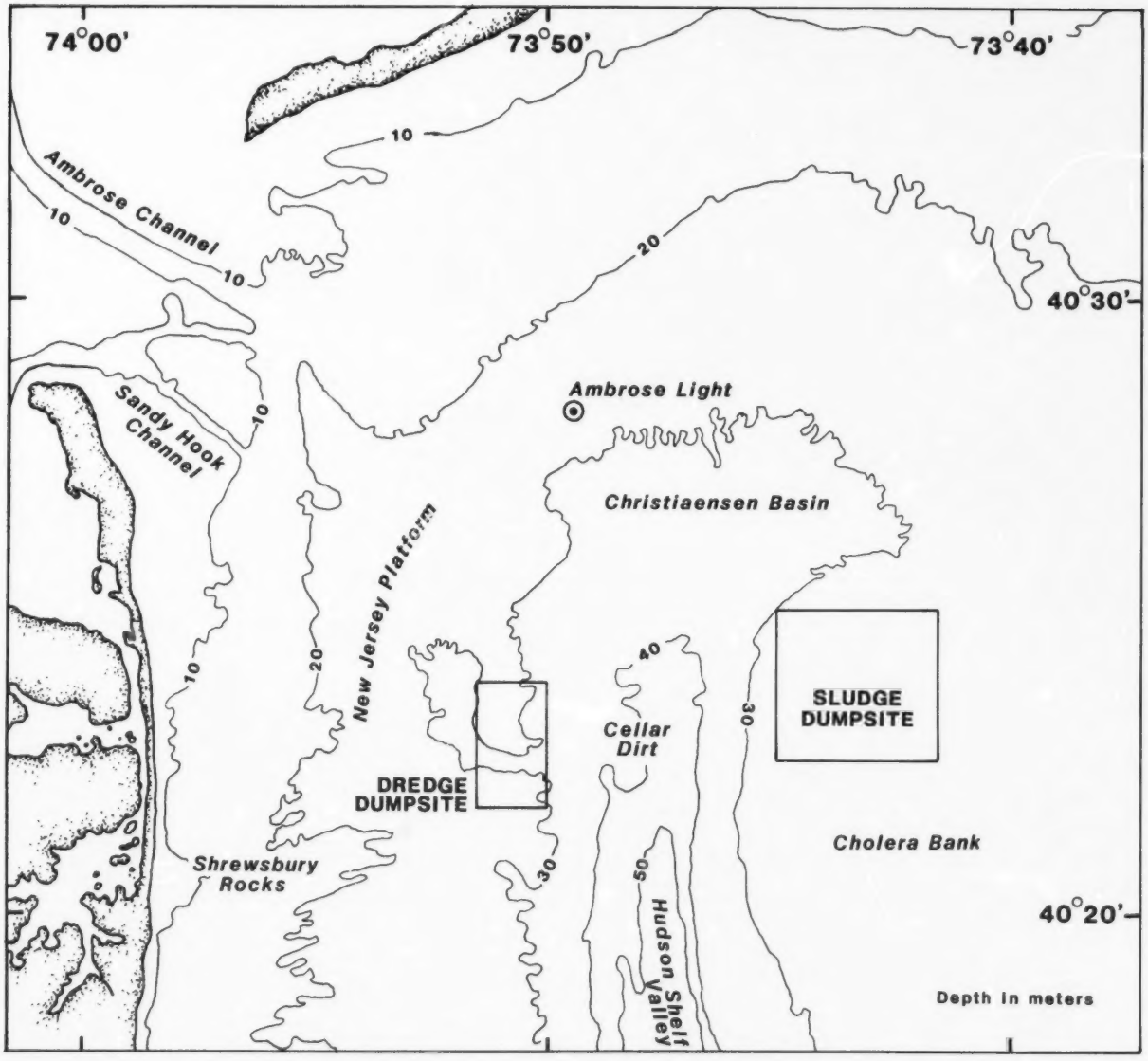


Figure 1. Location of the 12-mile sewage sludge dumpsite and dredge spoils dumpsite in the New York Bight.

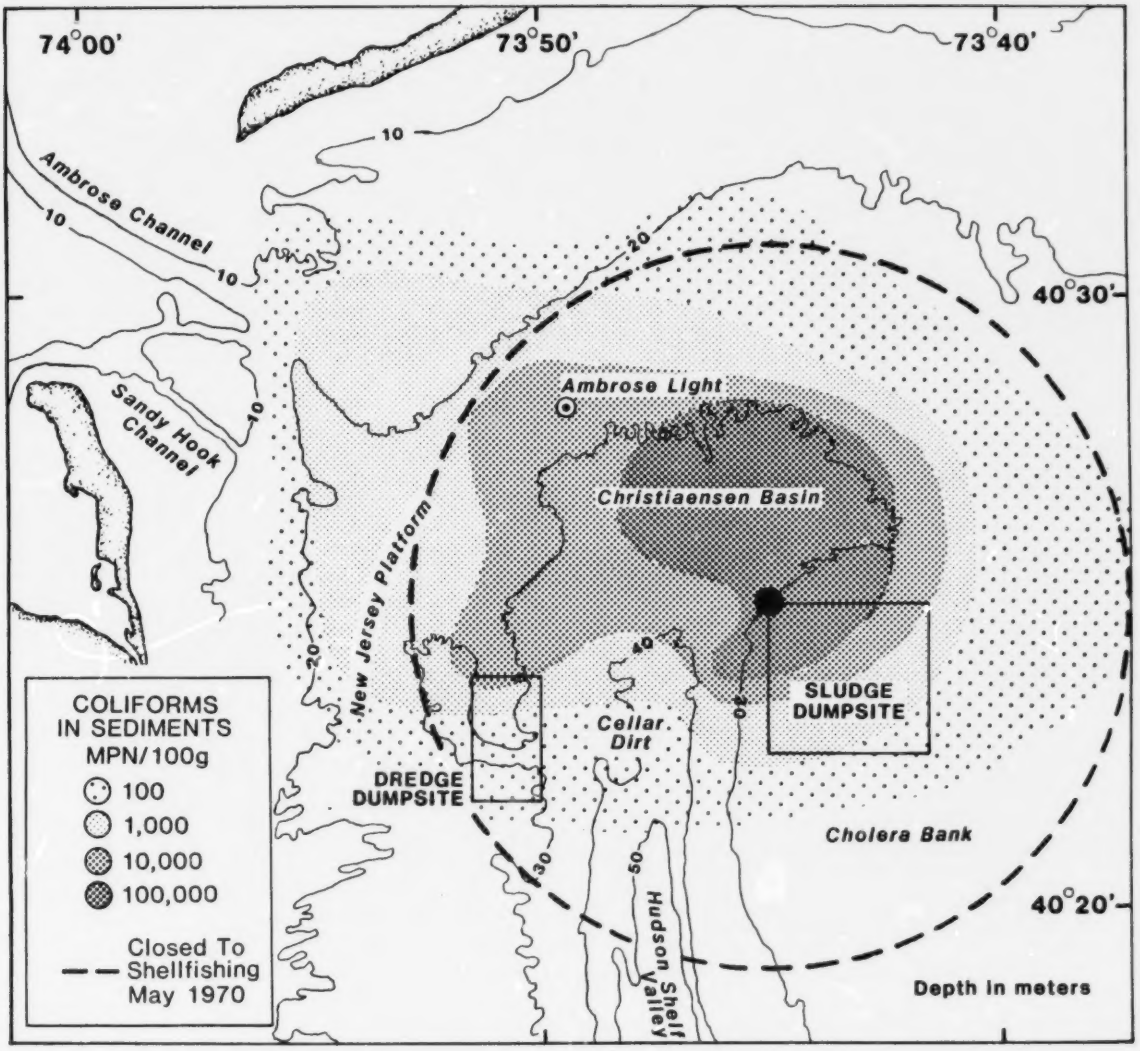


Figure 2. Coliforms in sediments and area closed to commercial shellfishing in the New York Bight in 1970 (after Verber, 1976).

continuous use since 1924, has served as a repository for sewage sludge from as many as 200 sewage treatment plants in the New York-New-Jersey area. Although the number of municipalities which presently use this site has decreased to nine, the volume of sludge dumped has increased as treatment facilities have been upgraded. In 1974, approximately 4.2×10^6 wet tons were dumped while by 1983, the volume had almost doubled, reaching 8.3×10^6 wet tons (Fig. 3; Santoro, 1987).

In April, 1985, following lawsuits by both New York and New Jersey, EPA Region II effectively closed the 12-mile dump site by denying requests for its redesignation (U.S. EPA, 1985). The selection of the 106-Mile Deepwater Municipal Sludge Disposal Site as the new disposal area necessitated a plan by which the dumpers would have adequate time to develop barging and hauling capability and budget the additional costs - estimated at approximately 20KK per year (Leschine and Broadus, 1985). To accommodate these changes, EPA adopted a schedule whereby the nine municipalities would begin to phase out dumping in a step-wise fashion in 1986 with all dumping to be discontinued by December 15, 1987 (Santoro, 1987). Thus, the abatement of dumping at the 12-mile site has presented a unique opportunity to determine the response of this ecosystem to removal of a major waste loading.

Little is known about the "recovery" of old dump sites. Historically, the only information documenting changes of this nature comes from surveys conducted before and after closure of the Philadelphia dump site (Devine and Simpson, 1985). In contrast with the depositional characteristics of the 12-mile dump site, the Philadelphia site was in a highly dispersive system which precluded any substantial sludge accumulation. Although some changes in benthic macrofauna had

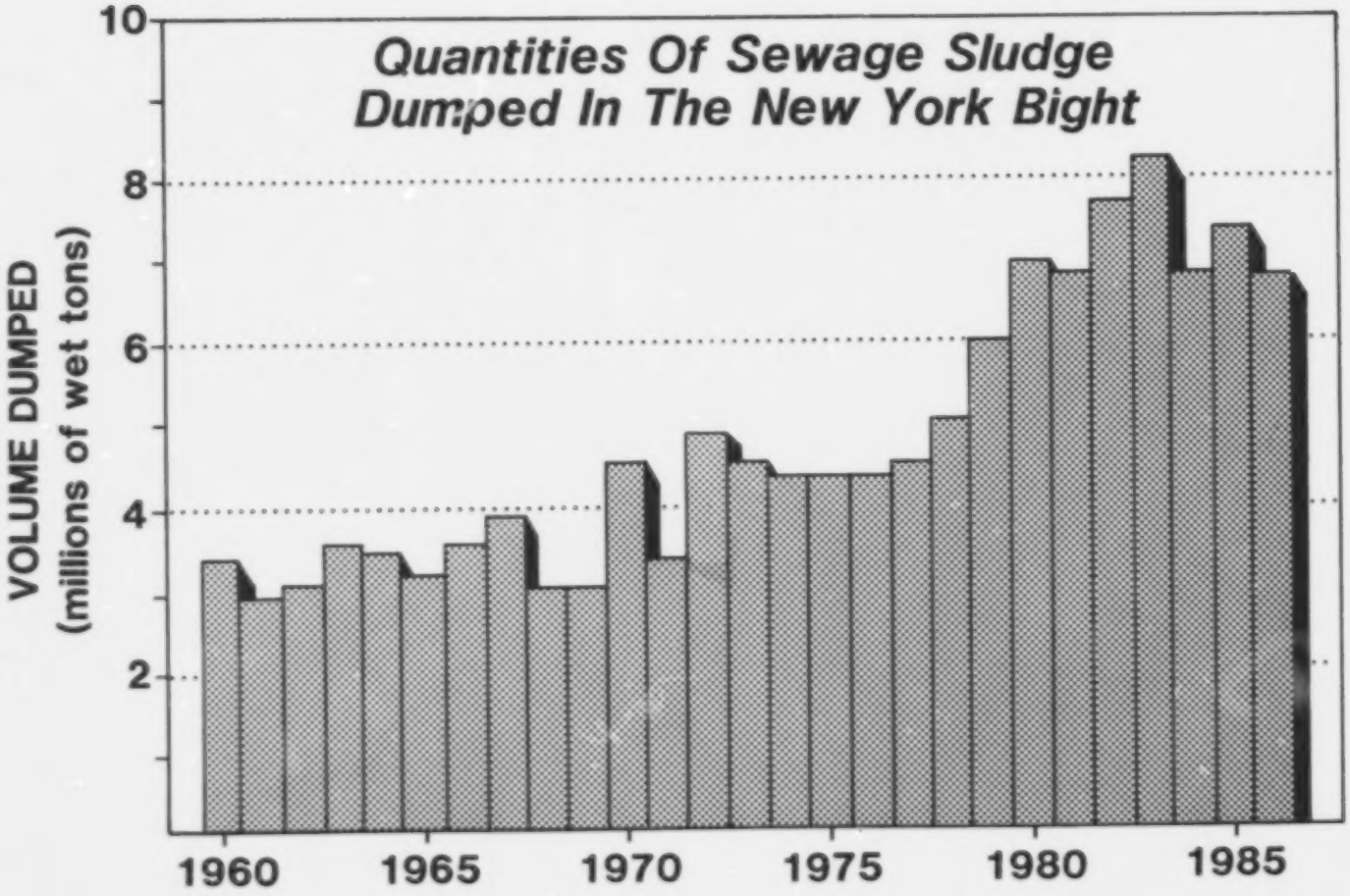


Figure 3. Estimated volumes of sewage sludge dumped in the New York Bight (after Swanson et al. 1985; Santoro, 1987).

occurred (Lear and O'Malley, 1983) which might reflect environmental degradation (Boesch, 1982), they were considered relatively minor. Nearly four years after the site was closed, pollution indicators had decreased and microbial contamination returned to background levels. This raises the question of whether abatement of dumping at the 12-mile dump site would eventually result in significant changes in water and sediment chemistry, microbial levels, benthic community structure, distribution, abundance and utilization by demersal finfish, or other ecological effects attributed to sewage sludge (Swanson et al., 1985). Since it is probable that waste dumping in shelf areas will be among options considered in the future, the Environmental Processes Division (EPD) of the Northeast Fisheries Center (NEFC), National Marine Fisheries Service (NMFS), National Oceanic and Atmospheric Administration (NOAA), has developed and initiated an interdisciplinary study in which biological, chemical and physical oceanographic approaches are integrated to document changes in living marine resources and their habitats during and following abatement (Environmental Processes Division, 1987a). Results should provide managers with the type of information which will allow better decision making as to whether increased costs of barging to offshore sites will be supported.

METHODS

Development of the Plan for Study

Several factors had to be considered in the development of a practical and efficient study plan. The first related to the long history of environmental degradation in the area. Separation of effects due to natural variability, as compared with those which are pollutant-

induced, is most effectively accomplished by comparing existing conditions prior to and following any change or by using comparable reference areas. In the case of the 12-mile site, there are no data characterizing the area before dumping began in 1924. Therefore, assessment of any changes in the habitat and biota will be based on conditions existing before dumping ceases. Past surveys of the New York Bight, e.g., NOAA's 1970s Marine Ecosystem Analysis Project (MESA) (for review, see Gross, 1976; Mayer, 1982) and 1980's Northeast Monitoring Program (NEMP) (U.S. NOAA, 1987), will provide some background data on sediment contaminants and benthic communities in the area. However, neither study incorporated an integrated sampling design in which a full range of physical, chemical and biological variables were measured systematically on a close spatial-temporal scale.

A second difficulty in determining direct causal relationships pertains to the separation of effects of pollutants from various sources. It is estimated that only 11% or less of the contaminant load in the New York Bight is traceable to sewage sludge; the major sources being via the Hudson-Raritan plume and dredged material which is dumped on the western edge of the Christiaensen Basin (Table 1; New York City Department of Environmental Protection, 1983). Finally, as has been pointed out (Capuzzo et al., 1985a), determining direct cause-effect relationships at the population and community levels is, in most instances, nearly impossible. Nevertheless, by conducting synoptic surveys and initiating the study in July 1986 before dumping was reduced, evaluation of the effects of abatement on a variety of marine biota and their habitats should be possible.

Table 1.

PERCENT DISTRIBUTION OF ESTIMATED POLLUTANT LOAD IN THE NEW YORK BIGHT[a]

Substance	Approximate total load ($mt\ y^{-1}$)	Source			
		Hudson-Raritan Estuary (%)	Dredge Spoil (%)	Sewage Sludge (%)	Other[b] (%)
Total Solids	7×10^6	25.6	65.5	2.8	6.1
Oil and Grease	206,000	62.0	15.4	8.1	14.5
Arsenic	134	54.6	43.1	2.1	0.2
Cadmium	105	55.0	31.3	8.4	5.3
Chromium	1,640	45.4	39.4	9.3	5.9
Copper	3,000	41.3	45.6	11.1	2.0
Lead	2,130	48.0	40.0	9.2	2.8
Mercury	53	53.2	30.6	3.1	13.1
Nickel	1,000	62.0	33.5	4.2	0.3
Zinc	6,840	50.1	38.6	8.1	3.2
PCBs	9.7	47.4	47.9	3.1	2.1
Fecal Coliforms	Most probable number ($MPN\ y^{-1}$)				
	4.84×10^{19}	99.0	No Data	0.2	<0.001

(a) Adapted from N.Y. City DEP, 1983.

(b) other includes acid waste, cellar dirt, atmosphere, N.J. and L.I. coastal discharge.

Assuming that introduction of sludge affects, and is affected by a wide range of physical, chemical and biological factors, a number of working hypotheses were developed. Regarding fishery resources, for example, while dumping continues, abundance, distribution and species composition of finfish and benthic communities will differ significantly among areas which are bathymetrically similar but represent a gradient of sludge influence ranging from heavily polluted to "clean" sites. Following cessation of dumping and predictable shifts in sediment contaminants and benthic forage species, these differences in abundance, distribution and composition should be reduced among sites. Correlated with these changes, diets of demersal finfish and larger invertebrates, i.e. decapod crustaceans, would be modified to reflect shifts in benthic prey availability. Significant reductions should also occur in microbial concentrations. Coliform bacteria in surf clams should decrease to acceptable levels and permit shellfish beds to be reopened.

Changes in water and sediment chemistry will also be evident. Sediment trace metals in sludge depositional areas should be reduced although certain compounds, e.g., PCB concentrations, will remain relatively unchanged (Table 1). Seabed oxygen consumption and nutrient regeneration rates reflective of benthic community metabolism will fall to more natural levels. Dispersion of sewage sludge and subsequent cleansing of the Christiaensen Basin will be influenced by local windfield conditions and stratification of the water column. Based on these hypotheses, a range of variables was selected to be measured based on their relevance to fishery resources and as accepted indicators for detecting environmental changes (Table 2).

Table 2.

12-MILE DUMP SITE VARIABLES MEASURED

Habitat

WATER

- **Bottom water**
  - Dissolved oxygen (R,B)
  - Temperature (R,B)
  - Salinity (R,B)
  - pH (R,B)
  - Sulfide (R,B)
  - Ammonium (R,B)
  - Turbidity (R,B)
- **Water Column**
  - Temperature, Salinity, Oxygen } (CTD)
  - Current measurements (moored current meters)

SEDIMENTS

- **Chemistry**
  - Organic contaminants (R,B)
  - Sulfide, pH profiles (R)
  - Redox potential (R,B)
  - Sediment BoD (R)
  - Chlorophyll pigments (R)
  - Total organic carbon (R,B)
  - Heavy metals (R,B)
- **Characteristics**
  - Grain size (R,B)
  - Erodibility (R)
- **Rates**
  - Seabed oxygen consumption
  - Sedimentation

R=Replicate Survey

B=Broadscale Survey

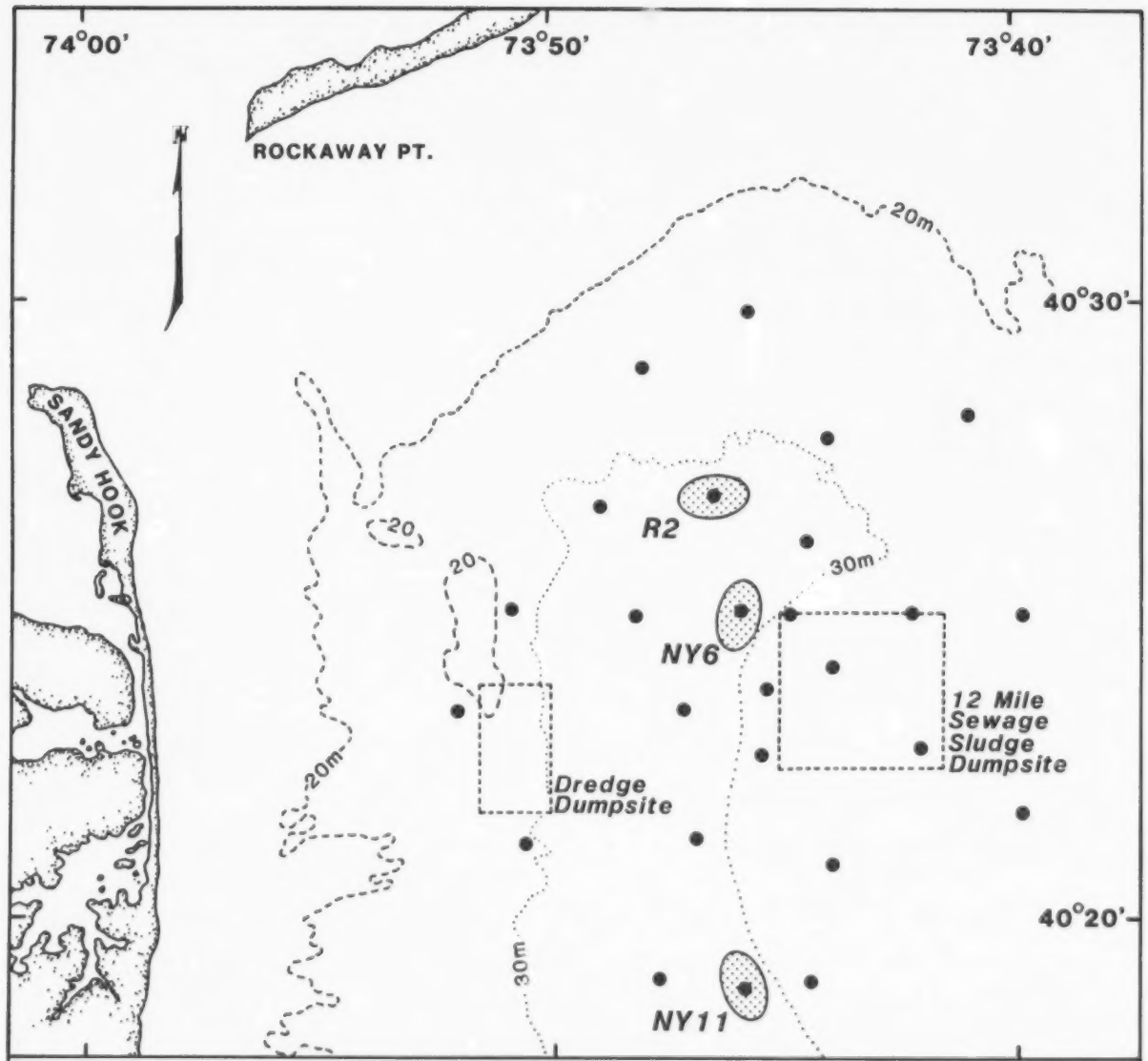
Biota

BOTTOM COMMUNITIES

- **Fish**
  - Distribution/abundance (R,B)
  - Diet (R)
    - winter flounder
    - red hake
    - silver hake
    - lobster
  - Gross pathology (R,B)
  - Tissue organics (R)
    - winter flounder
    - lobster
  - Migration(tagging) (B)
    - winter flounder
- **Benthos**
  - Macrofauna abundance/diversity (R,B)
  - Meiofauna abundance/diversity (R)
- **Bacteria-sediments**
  - Fecal and total coliform (R)
  - *C. perfringens* (R)
  - *Vibrio* spp. (R)
  - Epifluorescence count (R)
- **Bacteria-shellfish**

The study is developed around two complementary sampling series: (1) a Replicate survey and (2) a Broadscale survey, each conducted in alternate months (Fig. 4). In the former, eight replicate measurements for most variables are made at three stations for which historical data exist and which have different amounts of sewage sludge accumulation but similar bathymetry and sediment types. The most polluted, NY6, is located approximately 1.6 km down slope from the corner of the dump site where the heaviest dumping has occurred; it is expected to have the greatest accumulation of sludge and contaminants. Benthic macrofaunal populations in this area are characterized by the polychaete, Capitella capitata, found in sediments which are organically enriched and environmentally disturbed (Boesch, 1982; Steimle et al., 1982). A second station, R2, is about 3.4 km north of the dump site center on the north edge of the Christiaensen Basin. Benthic communities are presumably "enriched" with a high biomass of tolerant, though not necessarily pollutant-indicator species. The third replicate station, NY11, is nearly 10 km south of the dump site center on the eastern shoulder of the Hudson Shelf Valley. It is expected to be the least polluted of the three sites with the lowest concentration of sediment contaminants and characterized by benthic macrofaunal communities typical of Upper Shelf Valley sediments.

The broadscale survey consists of single, non-replicated bimonthly measurements at 23 stations covering most of the inner Bight and includes all major habitat types (Fig. 4). Station selection was based on considerations of bathymetry, known distribution of contaminants, patterns of environmental variables, dumping and dispersion patterns of sewage sludge and existing historical benthic data. Additional surveys



	1986						1987												1988					
	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D
Replicate ⊙	x	x	x		x		x		x		x		x	x	x		x		x		x		x		x	x	x		x	
Broadscale ●		x		x		x			x		x		x		x		x			x		x		x		x		x	
Seabed metabolism	x	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Hydrographic surveys (EPA, NJ DEP, FDA and others)										x			x									x			x				

Figure 4. Replicate and broadscale stations and scheduled surveys of the 12-mile dumpsite.

with EPA, FDA, New Jersey Department of Environmental Protection (N.J. DEP) and others will complement these efforts.

Data from the replicate surveys will be used to test hypotheses, develop models and evaluate effects of different contaminant levels; information from the broadscale surveys is expected to provide data describing changes in the spatial extent of sludge derived contaminants and correlation among variables.

RESULTS

Although only preliminary results are presently available, data from the first months of the replicate survey generally substantiate the selection of the three areas as representing a gradient of sludge influence. As expected, NY6 appears the most heavily altered and least stable while R2 reflects biotic characteristics associated with organic enrichment. The reference station, NY11, may be more transitional than anticipated with a mixture of fauna and sediments characteristic of both "clean" and disturbed areas.

Fish and Invertebrates

Abundance and distribution of fishes and invertebrates followed known patterns of seasonal occurrence and migration within the New York Bight (Grosslein and Azarovitz, 1982; Environmental Processes Division, 1987b). Based on the seven combined surveys of the three replicate stations from July 1986 through May 1987, 26 species of demersal finfish were collected with little skate, Raja erinacea, and winter flounder, Pseudopleuronectes americanus, numerically most abundant as well as comprising 59% and 12% respectively of the total demersal biomass. Other demersal species, ranked in order of abundance, included silver

hake, Merluccius bilinearis (3% of the total demersal biomass), red hake, Urophycis chuss (7%), and ocean pout, Macrozoarces americanus (9%). Although the survey was not designed to monitor semipelagic and pelagic fishes, ten species were collected with butterfish, Peprilus triacanthus, most abundant. Of the 13 species of invertebrates, nine were decapod crustaceans. Rock crab, Cancer irroratus, was the dominant species numerically as well as by weight, comprising 63% of the total invertebrate biomass.

From early fall through late winter, the largest biomass of fishes and invertebrates, primarily little skate and rock crabs, was caught at the enriched site, R2, but in late spring and summer biomass was highest at the most polluted station, NY6, due primarily to heavy concentrations of rock crabs (Fig. 5). On an annual basis, however, mean biomass for all species was more than two times as high at R2 ($\bar{x}$ = 24.5 kg tow^{-1}) as compared with NY6 ($\bar{x}$ = 11.3 kg tow^{-1}) or the cleaner, reference station, NY11 ($\bar{x}$ = 11.1 kg tow $^{-1}$). For demersal finfish the difference is even greater with the biomass over two and a half times as high at R2 ($\bar{x}$ = 21.0 kg tow $^{-1}$) than at NY6 ($\bar{x}$ = 7.9 kg tow $^{-1}$) and three times higher than NY11 ($\bar{x}$ = 6.9 kg tow^{-1}).

Incidence of disease was low, averaging 2% or less among winter flounder, red hake and silver hake. Higher incidence was usually associated with fish collected from R2, the enriched site.

Benthic macrofauna

Differences in benthic macrofauna also apparently support expected variation among the three sites although only data from the first summer is presently available. From July through September, 1986, the most heavily polluted area, NY6, had the lowest number of benthic species,

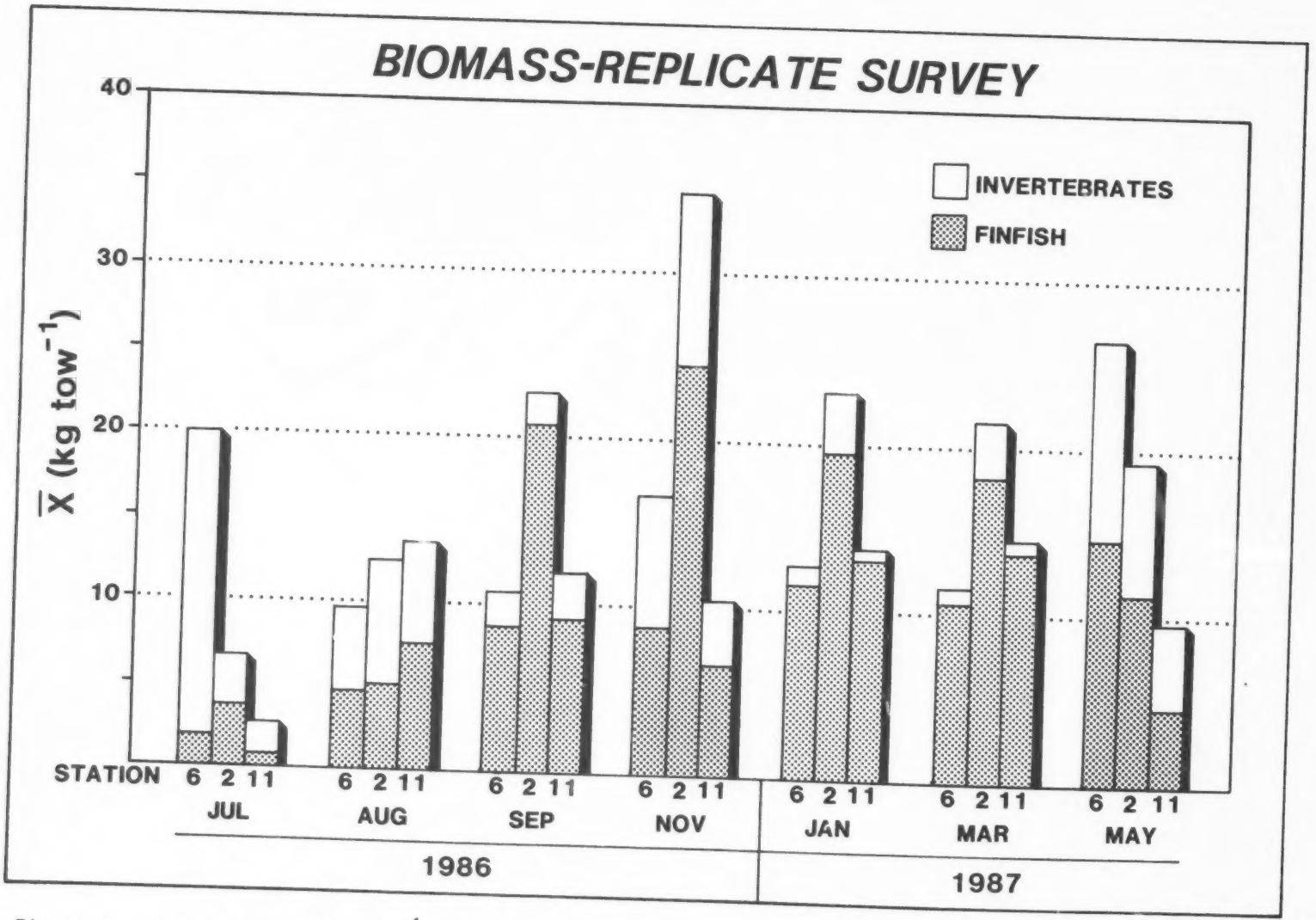


Figure 5. Mean biomass (kg tow^{-1}) of fishes and invertebrates collected during seven surveys of three replicate stations adjacent to the 12-mile dumpsite. (S. Wilk and A. Pacheco, pers. comm., 1987).

including pollution sensitive molluscs and crustaceans, while the cleaner reference site, NY11, had the highest number (Fig. 6, A,B,C; Environmental Processes Division, 1987b). Support for these findings comes from *in situ* experimental tray studies which indicate that significantly fewer numbers of benthic larvae settle in sediments contaminated with sewage sludge in comparison with controls (Environmental Processes Division, 1987b).

As predicted for NY6, *Capitella capitata* was the dominant species both in terms of numbers of individuals (Fig. 6D) as well as biomass in July, 1986, reflecting the disturbed nature of this area. Population density of this species at NY6 was one and four orders of magnitude greater than at NY11 and R2 respectively. However, the population crashed in August with little sign of recovery in September. The occurrence of *Capitella* at NY11, along with amphipods which are usually not found in disturbed areas, indicates this may be a more transitional zone than anticipated. This is supported by sediment analyses (see below) which show a layer of sludge-like material underlying surficial fine sand.

Bacteria

Analysis of sediments collected from the three replicate sites from September 1986 through February 1987, indicate that concentrations of bacteria (total and fecal coliform, *Escherichia coli*, *Vibrio* spp., and *Clostridium perfringens*) were generally highest at the most polluted site, NY6, less at the enriched station, R2, and lowest at NY11 (Environmental Processes Division, 1987b).

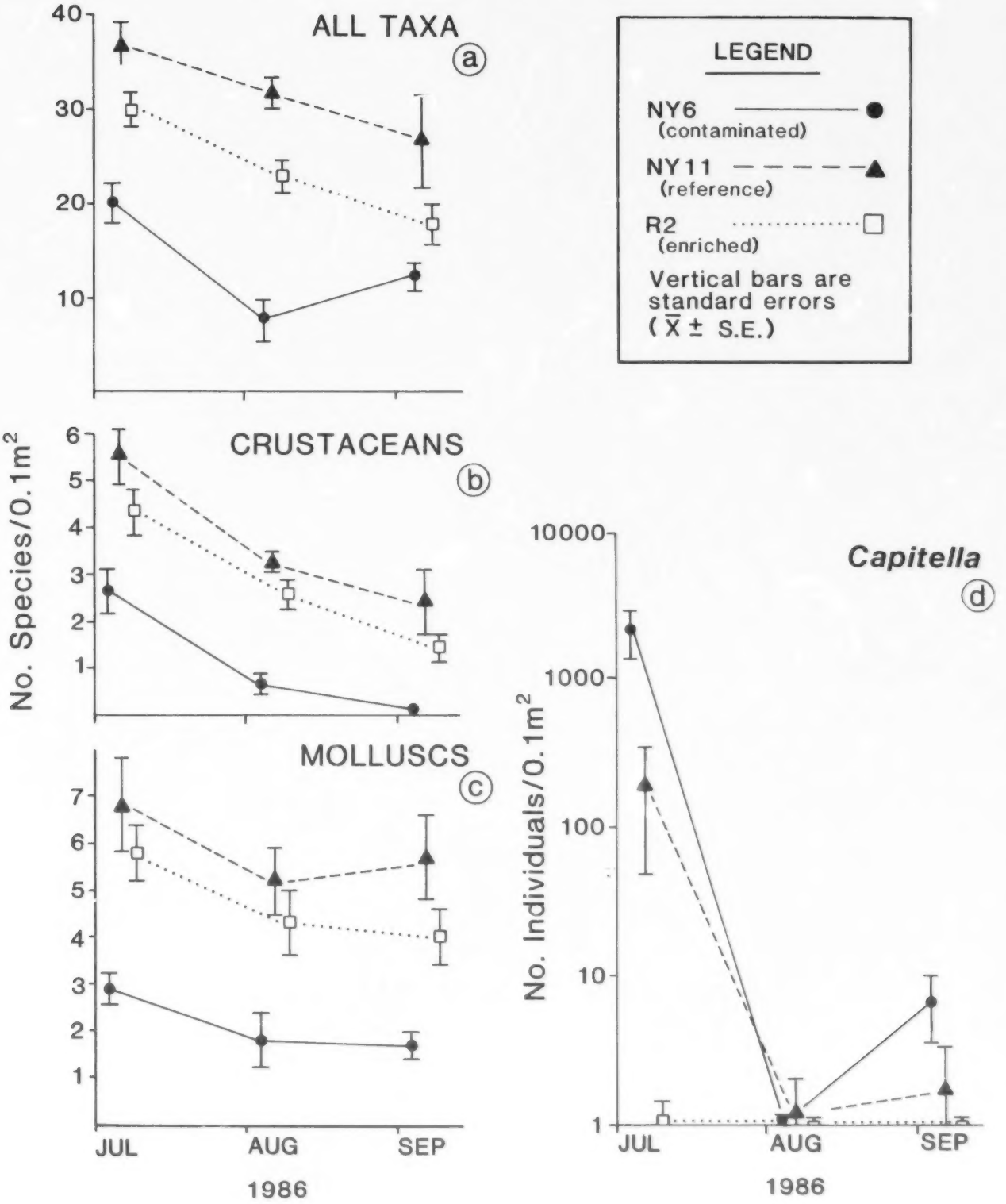


Figure 6. Changes in abundance of (a) macrobenthic species, (b) crustaceans, (c) molluscs, and (d) numbers of Capitella during July, August and September, 1986 at the 3 replicate stations. (R. Reid, pers. comm., 1987).

Water and Sediment Chemistry

Seasonal fluctuations characterize seabed oxygen consumption (SOC) in the New York Bight with rates higher in summer than in winter (Environmental Processes Division, 1987b). Typical fluctuations have been monitored since 1974 across a transect extending through the Christiaensen Basin and 12-mile dump site. In general, prior to any abatement of dumping, rates were lower in areas east of the dump site, increasing to the west in the organically enriched Christiaensen Basin (Fig. 7). Following the 30% reduction in dumping, SOC rates appear to be gradually decreasing to the east of the dump site. This may be due in part to the fact that this is close to the area where Nassau County barges once dumped (Fig. 7A). This decline is not evident in regions close to the western corner and the most polluted station, NY6, where dumping has continued at the same level (Fig. 7B,C).

A preliminary analysis of sediment redox potential at NY6 from July to September, 1986, indicates that fluctuations in reducing conditions were matched by the changes in numbers of Capitella (Environmental Processes Division, 1987b). For example, in July when the biomass was the largest (Fig. 6D) reducing conditions prevailed in the sediments; by August when Capitella had almost disappeared, oxidation conditions in the sediment increased.

Analysis of other baseline data related to water and sediment chemistry, e.g., total organic carbon, total plankton respiration, interstitial sulfide concentrations, heavy metals and organics, are presently in progress.

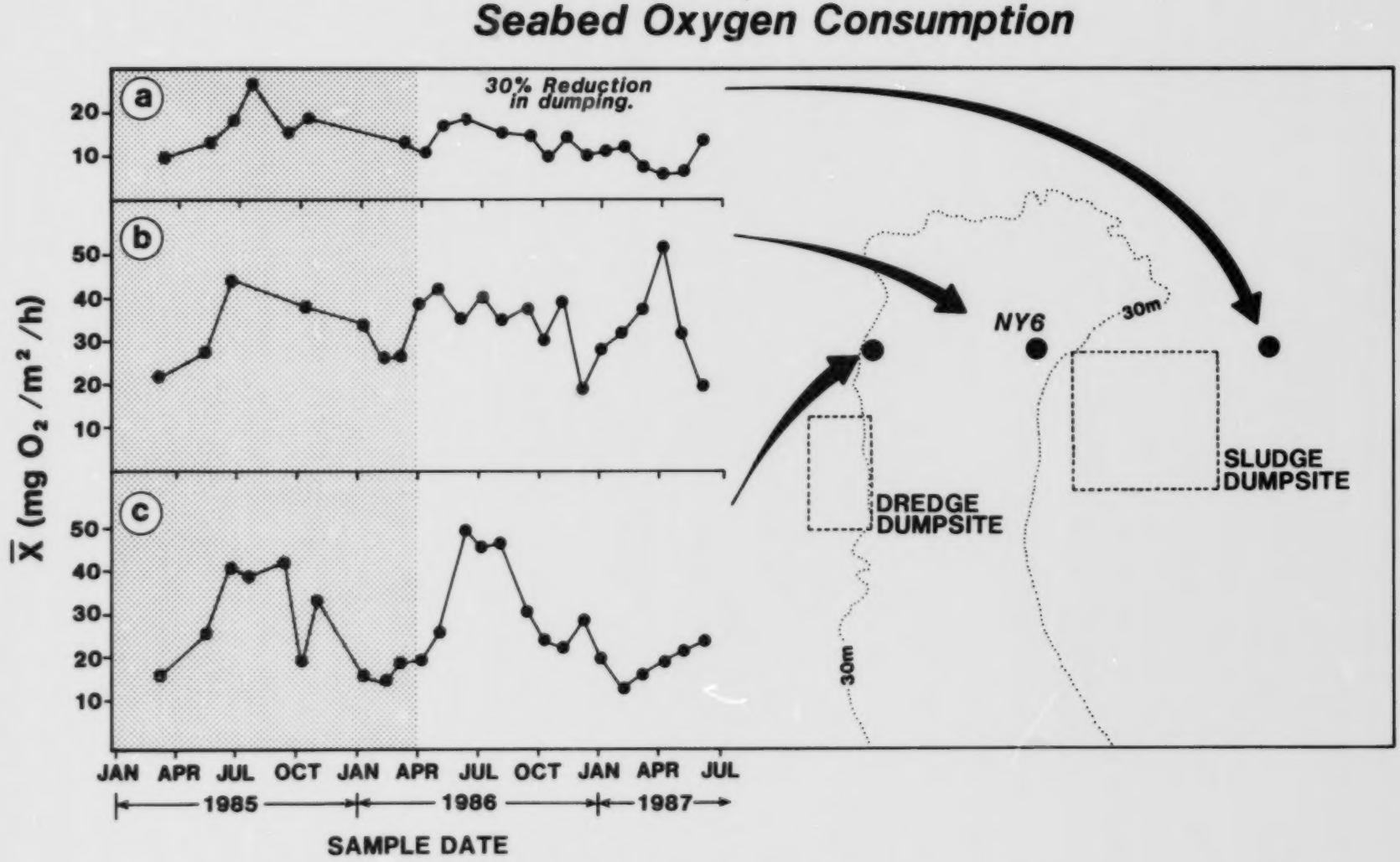


Figure 7. Seasonal fluctuations in seabed oxygen consumption prior to and following 30% reduction in dumping of sewage sludge at (a) site to the east of the 12-mile dumpsite, (b) replicate station, NY6, and (c) site to the west of the 12-mile dumpsite. (W. Phoel, pers. comm., 1987).

Sediment Resuspension/Circulation/Hydrography

Supporting the premise that sediment quality is the most altered at NY6, measurements of sediment resuspension confirmed that it was the least stable of the three replicate stations (Environmental Processes Division, 1987b). During surveys in October, 1986, NY6 sediments were characterized as well-mixed sand-silt with net resuspension 10 to 100 times greater than the aerobic well-mixed sand-silt collected at the enriched station, R2. The reference site, NY11 had well-mixed sand-silt with a fine sand interface and was more stable than NY6 but net resuspension of the sediment was about double that measured for R2. By March, 1987, although there was little change evident at R2, a thick layer of watery silt (suspected but not confirmed to be sludge) had accumulated at NY6 and net resuspension was even higher. Data from current meters deployed in and around the dump site and Hudson Shelf Valley will be integrated to develop a model for sediment transport and distribution.

Synoptic surveys, in and around the Hudson Shelf Valley in mid-May 1987, indicate the rapid rate at which near-bottom conditions may change (Environmental Processes Division, 1987b). A small area of oxygen depleted water (< 80% saturation) which was detected to the northwest of the dump site, expanded nearly fivefold over a 5-day period until it extended about 20 km to the south of the area. During the same surveys, down-valley flow of bottom water was observed. In addition, profiles of low salinity (< 31 ppt) water indicated that the Hudson-Raritan plume was up to 15 m thick and extended for nearly 25 km along the northern New Jersey coast.

CONCLUSION

Probably no single issue related to the use of nearshore coastal waters in the New York Bight has generated as much public concern in the last several years as has the ocean disposal of wastes. Recent beach closures due to high coliform counts, restricted consumption of certain fish and shellfish harvested in local waters, as well as physical evidence of plastics, hospital waste, tar balls and other debris washing ashore, has again raised the spectre of ocean pollution destroying the coast. Regardless of statements that sewage sludge effects are minimal and that dumping has not resulted in unreasonable degradation of the marine environment (New York City Department of Environmental Protection, 1983) the public's perception is that dumping sewage sludge at the 12-mile site is a major factor affecting water quality and utilization of marine resources. Predictions as to the effects of cessation of dumping have ranged from no significant or measurable improvement (New York City Department of Environmental Protection, 1983) to modest ecological improvement, including reductions in sediment pathogenic bacteria and viruses, over a limited geographical area (Swanson et al., 1985). Only, however, if there were comparable reductions in disposal of dredge spoils and other waste loading to the Bight could any large-scale and significant improvement be expected (Gunnerson et al., 1982).

In a recent review of future research needed to promote efficient management and utilization of nearshore areas, the point was made that although vast resources have been directed at the problem, many decisions have been based on predictions from models which have been tested only on a limited scale (Capuzzo et al., 1985b). To confirm

these models, future efforts must focus on adopting an integrated analytical approach with a long-term commitment to understanding basic ecosystem processes. The present study, undertaken by the Northeast Fisheries Center, should begin to demonstrate the feasibility of such an approach.

ACKNOWLEDGMENTS

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SELECTION OF AN OCEAN DUMPSITE IN THE CANADIAN BEAUFORT SEA

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1. Introduction

In 1982, the Northwest Territories District Office of Environmental Protection initiated a program to investigate the feasibility of selecting one or more common-user dumpsites in the Canadian Beaufort Sea for the disposal of scrap metal and other inert, bulky wastes. This ocean dumpsite selection program was developed in response to requests from the oil and gas industry for approval to dump materials including scrap metal, rubber tires, concrete rubble, supply vessels, and drilling caissons into the ocean. Transportation costs associated with shipping these materials out of the Canadian Arctic make the ocean disposal option attractive to industry. Area residents have, however, been concerned about the Beaufort Sea being used as a repository for such wastes. These concerns are particularly relevant because the Beaufort Sea area is subject to a native land claim agreement, the Inuvialuit Final Agreement (Indian and Northern Affairs Canada, 1985), which requires that the Government of Canada consult with the claim holder, the Inuvialuit Regional Corporation, before approving activities within the agreement area. The ocean dumpsite selection process is intended to ensure that before any disposal of scrap metal and other bulky inert wastes is approved, the ocean dumping option is fully evaluated and a common-user site is identified. An objective of this strategy has been to avoid a proliferation of small dumpsites established on the basis of the immediate needs of individual operators. The purpose of this paper is to describe the process which was followed and to summarize the results and recommendations.

2. Volume of Wastes

The program was initiated with a report outlining the nature of the problem (M.M. Dillon Ltd., 1982), and a subsequent government/industry workshop to discuss issues and explore possible solutions (Arctic Laboratories Ltd., 1984). Workshop participants estimated that 375,000 tonnes of bulky inert wastes could be generated by petroleum and related industries over a 75 year period, and this value was increased by an order of magnitude, to 3,750,000 tonnes to allow a safety margin to account

for underestimates. It was projected that 1.0 km^2 would be covered with waste, on the assumption that the bulk density of the material would be 2 tonnes/m^3 and that the material would accumulate to a depth of 2 metres. This calculation did not, however, include the disposal of any large vessels or drilling caissons.

3. Constraints Affecting Site Selection

An analysis was carried out to identify and examine constraints which would affect the siting of an ocean dumpsite (Environmental Protection, 1986 and 1987). The factors which were identified included: the presence of biologically sensitive areas; proximity to resource harvesting activities; water depth; sediment stability; ice scour; ice as it affects access to the site; currents; legislation and policy; proximity to potential oil and gas development sites; and the costs of transporting wastes to the disposal site. The major constraints affecting site selection are depicted in Figures 1 and 2.

It was concluded that the effects of ocean disposal on some populations and biological communities would be greater if waste disposal areas were located in nearshore environments (<20 m) although the overall degree of impact would still be negligible in terms of regional biological resources. Marine mammals and fish are the most valued ecosystem components which could interact with ocean disposal activities. White whales (beluga) concentrate in areas off the Mackenzie Delta in the summer months and are harvested in a domestic fishery by native people from surrounding communities (Figure 1). Interactions with white whales can be avoided for the most part by ensuring that dumping does not occur in the Delta area. Bowhead whales tend to remain further offshore feeding primarily on near surface zooplankton although zooplankton from the nepheloid layer, and benthic organisms, may also be consumed. The distribution of bowheads tends to be more variable than beluga with the largest concentrations occurring along the Yukon coast during the last few years (Harwood and Ford, 1983; Harwood and Borstad, 1985; Duval et al., 1986). Bowhead whales are highly valued, particularly because they are classed as an endangered species; however, effects from dumping are expected to be limited to brief interactions during the actual process of dumping, however these interactions will most likely be followed by an avoidance reaction.

The redistribution of wastes by ice scour and the degree to which ice limits access to a disposal site were identified as constraints to site selection. Ice scour occurs in the nearshore zone to a water depth of 50 metres (Dome et al., 1982) and could re-distribute bulky inert wastes. Ice cover in the summer months increases further offshore as the polar pack ice is approached (Figure 2) and can present a navigational restriction to site access. It was concluded that re-distribution of material by ice scour can be avoided by locating the site where water depths exceed 50 metres and that areas are available north of the 50

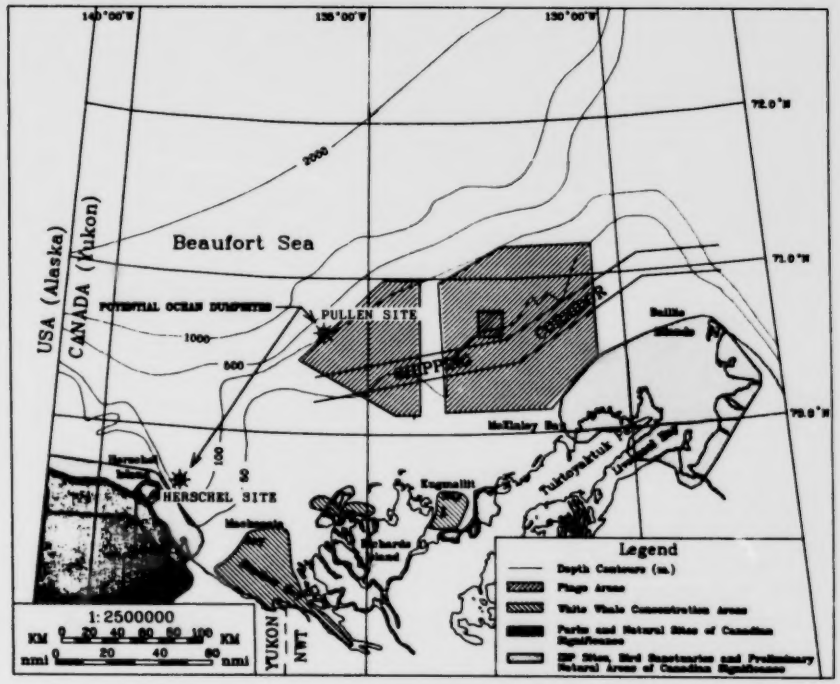

Figure 1. Constraints on ocean dumpsite selection (After Dickins et al.,1987).

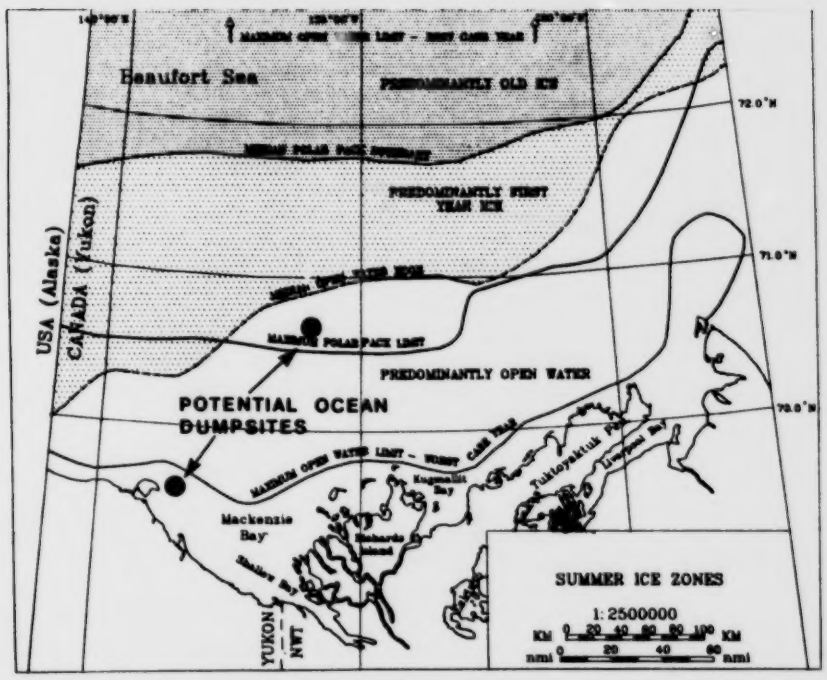

Figure 2. Summer ice zones (After Dickins et al.,1987).

metre contour but south of areas where pack ice becomes a constraint to navigation.

Water depth was considered to be a constraint in that it would be undesirable to locate a site where the dumped material could create a hazard to navigation. The Navigable Waters Protection Act stipulates that there must be 37 metres of clearance over structures placed on the seafloor. To allow for the dumping of drilling caissons or ships, it would therefore be necessary to ensure that the depth at the site exceeds 60 metres (Environmental Protection, 1986).

It was considered desirable, although not essential, that sediments be stable at the selected site. In nearshore areas (<50 m) sediment can be quite mobile due to current, wave and ice dynamics (Fissel and Birch, 1984) while at the continental shelf break, in areas where steep slopes exist, submarine slumping can occur (Dillon, 1982). It was concluded that sites should be located in water depths greater than 50 metres and that particularly steep slopes, where there is evidence of recent slumping, should be avoided.

In offshore areas summer bottom currents have maximum velocities ranging from 16 to 29 cm/sec while closer inshore, maximum velocities range from 40 to 50 cm/sec (Fissel and Birch, 1984). In winter, current velocities are lower due to the decoupling of winds and currents as a result of ice cover. It is not anticipated that these currents would be strong enough to displace bulky, inert material, although uncrushed oil drums and barrels could be rolled during extreme current events.

It is important that the disposal site is not located in areas which would interfere with oil companies' plans to explore for, or produce hydrocarbons in the future. Through consultation with industry, it was possible to ensure that potential dumpsites would not constrain future oil and gas development activities.

Certain locations in the Beaufort Sea area have been or might be designated as national or territorial parks, national areas or sites of Canadian significance, or areas of unique significance under the International Biological Program (Figure 1). It was necessary to avoid these areas when selecting preferred sites for ocean disposal.

Transportation costs are a constraint and were calculated on the assumption that the wastes would be loaded directly onto an oilfield supply vessel or onto a barge. It was estimated that one day would be required to load a ship or barge and three hours would be required for each 50 km voyage segment. From this a base cost of $21,000 per load, plus an additional $7,000 for each 50 km voyage segment, was calculated and factored into the overall site selection process in an attempt to minimize costs to the greatest extent possible.

4. Site Selection

Once the constraints affecting site selection were identified they were incorporated intc an analysis of preferred locations using a grid system combined with a rating scheme (Environmental Protection, 1986). As a result of this analysis, two locations were identified as preferred dumpsites: Site A (Pullen Site) 70° 39'N by 135° 50'W; and, Site B (Herschel Site) 69° 40'N by 138° 30'W (Figures 1 and 2). After consultation with other government departments, industry, native groups and the public, a decision was taken to proceed with detailed characterization of these sites (Environmental Protection, 1987).

5. Characterization of Potential Ocean Dumpsites

The seabed at two potential dumpsites was characterized in late August, 1986 by side-scan sonar coverage, depth sounding, and sub-bottom profiling. Surficial sediments were analyzed to determine the particle size distribution, and the trace metal and hydrocarbon content, and to characterize the benthic infaunal species and their density. Epibenthic communities and demersal fish were qualitatively assessed (Arctic Laboratories and LGL, 1987).

A total of 30 stations were sampled in and around each dumpsite. Sample stations were chosen randomly with 10 sites located within a 1.5 km radius of the dumpsite centre and 20 sites in the area between 1.5 and 3.0 km from the centre. At each station two grab samples were obtained using a Smith-MacIntyre grab sampler (0.1 m^2 bite). One grab sample was sieved for macrobenthos and the second was subsampled for trace metal, hydrocarbon, and particle size analysis. In addition, 5 separate trawls were conducted at the Pullen Site and four at the Herschel Site using a 10 metre Otter trawl, with a 2 cm mesh size, to qualitatively assess epibenthic communities and demersal fish.

5.1 Pullen Dumpsite (Arctic Laboratories and LGL, 1987)

The Pullen Dumpsite bathymetry was characterized as dipping gently toward the north-northwest at a slope of approximately 1 in 37 with the water depths ranging from 120 to 300 metres. Using side-scan sonar the seabed was characterized as being smooth and regular with only very occasional and scattered features being noted. These features consisted of a single pock-mark which probably derived from shallow gas venting, and numerous shallow trenches 5 to 15 metres in length, which were not associated with any known geological phenomena. Within the shallow sub-surface sediments of the site area, the acoustic records showed a thick sequence of conformable soft marine sediments. These sediments ranged from 38 to 60 metres in thickness and were well banded suggesting thin layers of silts and clays similar to the type of materials recovered in grab sampling.

Results of the grab sampling and analysis program indicated that the top 2 cm of sediment was composed of predominantly clay size particles (overall mean 62.7% by weight) with less than 0.2% by weight of that layer being composed of sand or larger size particles. With the exception of mercury, concentrations of metals and hydrocarbons in surface sediments were uniformly distributed within the dumpsite and as far as 3 km from the dumpsite centre. Mercury concentrations were higher and more variable within the dumpsite as compared to the surrounding area. In general, the concentrations of all parameters were low and typical of pristine marine sediment (Tables 1 and 2).

TABLE 1. MEAN SEDIMENT TRACE METAL CONCENTRATIONS AT TWO POTENTIAL BEAUFORT SEA OCEAN DUMPSITES.

TRACE METALS PULLEN SITE
(ug/g; n=20)

	$\bar{x}$	S.D.
Lead	23.4	2.7
Nickel	36.5	3.9
Mercury	0.072	0.014
Copper	34.1	3.2
Zinc	136	13
Cadmium	0.1	0.02
Chromium	131	10
Barium	864	35

TRACE METALS HERSCHEL SITE
(ug/g; n=20)

	$\bar{x}$	S.D.
Lead	21.2	2.2
Nickel	33.8	4.3
Mercury	0.066	0.010
Copper	30.9	1.9
Zinc	121	10
Cadmium	0.11	0.02
Chromium	121	11
Barium	856	31

TABLE 2. MEAN SEDIMENT HYDROCARBON CONCENTRATIONS AT TWO POTENTIAL BEAUFORT SEA OCEAN DUMPSITES.

HYDROCARBONS-PULLEN SITE (ng/g)

	$\bar{x}$	S.D.	Range	n
Total Alkanes	6400	1800	4800-9200	19
Total PAH	660	78	430-750	18

HYDROCARBONS-HERSCHEL SITE (ng/g)

	$\bar{x}$	S.D.	Range	n
Total Alkanes	5000	1300	3000-8300	20
Total PAH	540	56	440-630	14

The infaunal characterization indicated that the average abundance was 3,474 organisms/m^2 with a mean biomass of 22.73 g/m^2 (Table 3). Polychaetes were the dominant taxa accounting for 67% of the total number of organisms and 57.5% of the total biomass. Tharyx sp. was the dominant polychaete taxon in terms of numbers while Lumbrinereis minuta, Prionospio sp., and Aricidea nr. lopezi were also abundant (Table 4). Other dominant groups included bivalves (Thyasira gouldii) and brittle stars (Ophiacantha bidentata)(Tables 3 and 4). In general, the infauna were typical of a featureless undisturbed mud bottom.

TABLE 3. Group composition of benthos collected in grab samples at 147-275 m depths at two potential dumpsites in the Canadian Beaufort Sea during August, 1986 (Arctic Laboratories and LGL, 1987).

Taxon	% of Total			
	Numbers		Biomass	
	Pullen Site	Herschel Site	Pullen Site	Herschel Site
Cnidaria				
Alcyonacea	0.12	0.03	0.21	4.46
Actiniaria	3.51	0.11	2.67	1.00
Nemertinea	1.09	1.11	2.36	1.78
Polychaeta	66.75	66.51	57.42	31.29
Mollusca				
Aplacophora	0.98	0.31	1.68	0.29
Gastropoda	0.20	0.11	1.24	0.43
Bivalvia	6.79	9.23	7.43	44.94
Sipunculida	0.83	0.55	0.35	0.59
Crustacea				
Ostracoda	5.30	12.87	0.83	0.44
Cumacea	4.92	2.39	0.25	0.10
Tanaidacea	2.01	2.74	0.11	0.05
Isopoda	0.86	0.21	0.36	0.15
Amphipoda	3.94	2.76	0.83	0.64
Decapoda	0	0.02	0	5.29
Echinodermata				
Asteroidea	0.03	0.02	7.45	6.38
Ophiuroidea	1.04	0.79	15.86	2.07
Ascidiacea	1.61	0.24	0.92	0.10
Total number (no./m^2) or biomass (g/m^2)	3474	6199	22.83	91.47

Table 4. Densities (no./m^2) of dominant species or genera of benthos collected in grab samples at 147-275 m depths at two potential dumpsites in the Canadian Beaufort Sea during August, 1986 (Arctic Laboratories and LGL, 1987).

Taxon	Pullen Site Mean ± SD (n = 10)	Herschel Site Mean ± SD (n = 10)
Tharyx sp.	381 ± 106	347 ± 156
Tauberia gracilis	18 ± 13	659 ± 252
Lumbrinereis minuta	257 ± 126	393 ± 119
Prionospio spp.	282 ± 227	101 ± 49
Maldane sarsi	52 ± 40	304 ± 109
Pholoe minuta	57 ± 39	306 ± 96
Aricidea nr. lopezi	238 ± 107	88 ± 55
Terebellides stroemi	69 ± 62	219 ± 65
Aricidea quadrilobata	159 ± 50	100 ± 49
Thyasira gouldii	154 ± 87	96 ± 50

Fauna obtained in the trawl survey were also typical of a featureless mud bottom. Notable taxa included Arctic cod (Boreogadus saida), starfish Hymenaster pellicudus; Pontaster tenuispinus), basket stars (Gorgonocephalus arcticus), brittle stars (Ophiacantha bidentata; Ophiopleura borealis), shrimp (Eualus gaimardi; Sabinea septemcarinata), five species of amphipods, four species of pycnogonids, bivalves (Astarte crenata), and sea anemones (Metridium senile). The trawl data confirmed that organisms which could exploit the increased habitat opportunities created by bulky inert material were present at the site.

5.2 Herschel Dumpsite (Arctic Laboratories and LGL, 1987)

The Herschel Dumpsite bathymetry was characterized as dipping gently toward the north at a slope of approximately 1 in 360 with water depths ranging from 137 to 150 metres. Using side-scan sonar, the seabed was characterized as being smooth and regular with very occasional and scattered debris. Some of the debris appeared to be sunken logs or branches that had drifted to the region via the Mackenzie River. Other features which were observed consisted of shallow trenches 5 to 15 metres long and similar to those observed at the Pullen site but somewhat more numerous. Sub-seabed conditions were characterized by a finely layered conformable soft sedimentary cover of at least 25 to 35 metres thickness over the entire site with no unusual features.

Results of the grab sampling and analysis program indicated that the 2 cm surface layer of sediment was a mixture of clay, silt, and sand sized particles (50, 40, and 10% by weight respectively). Concentrations of metals and hydrocarbons in surface sediments were uniformly distributed within the dumpsite and as far as 3 km from the dumpsite centre. In general the concentrations were low and typical of pristine marine sediment (Tables 1 and 2).

The infaunal characterization indicated that the average abundance and biomass was considerably greater than at the Pullen site, being 6199 organisms/m^2 and 91.47 g/m^2 respectively. Polychaetes were again dominant comprising 57.42% of the number of organisms and 31.29% of the biomass. Dominant polychaete species included Tauberia gracilis, Lumbrinereis minuta, Tharyx sp., Pholoe minuta, and Maldane sarsi (Table 4). The bivalve species Bathyarca glacialis was noted as being particularly common as were the species Thyasira gouldii and Nuculana pernula. In general the infauna at the Herschel Site were more abundant and diverse than at the Pullen Site.

Fauna obtained in the trawl survey were similar to the Pullen site but were qualitatively assessed as being more abundant and diverse. Notable taxa included arctic cod (Boreogadus saida), starfish (Ctenodiscus crispatus; Pontaster tenuispinus; Urasterias lincki), basket stars (Gorgonocephalus arcticus), brittle stars (Ohiacantha bidentata; Ophiocten sericeum), bivalves (Arctinula greenlandica; Boreogadus glacialis; Nuculana pernula), four species of pycnogonids, nine species of amphipods, shrimp (Eualus gaimardi; Sabinea septemcarinata), and sea anemones (Metridium senile). The trawl data confirmed that organisms were present which could benefit from the habitat opportunities offered by inert bulkly material.

6. Current Status of Dumpsites

Decisions on the location of disposal sites are made in response to specific applications; however, the ocean dumping program is managed to ensure that where possible and appropriate dumping is confined to a relatively small area. Environmental Protection has prepared documents which allow for the rational selection of preferred areas for locating multi-user dumpsites for bulky inert wastes. Within these areas two sites have been selected and, physically, chemically, and biologically characterized. The Arctic Regional Ocean Dumping Advisory Committee has concluded that one centrally located site would be preferred, and of the two sites characterized, attention will be directed toward the Pullen Site when future ocean dumping permit applications are considered.

There is no provision for the formal designation of an ocean dumpsite under the Ocean Dumping Control Act. When Ocean Dumping Permit applications are received they will be screened by Environmental Protection to ensure compliance with Schedules I, II

and III of the Ocean Dumping Control Act. In addition to the established Ocean Dumping Advisory Committee consultation process, the applications will be forwarded to the Inuvialuit Environmental Impact Screening Committee as specified in the Inuvialuit Final Agreement (Indian and Northern Affairs Canada, 1985). Decisions on disposal at the recommended site will be made on an application-by-application basis.

Acknowledgements

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RESULTS OF THE SEABED WORKING GROUP ASSESSMENT OF SUBSEABED DISPOSAL OF HIGH-LEVEL RADIOACTIVE WASTES

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Since 1977 a group of nations have cooperated through the Seabed Working Group (SWG) of the Nuclear Energy Agency to conduct research on ocean floor burial of high-level radioactive wastes. There is now real and appropriate data to support each of the steps leading to dose calculations, geologic characterization of a few promising locations, and the demonstration of emplacement techniques. The SWG is preparing to publish their results in a series of volumes in late 1987.

Implantation of wastes has been demonstrated to be at least theoretically feasible for both penetrator and drilled techniques. Techniques to evaluate the long term geological processes relevant to the containment in deep-sea sediments are available. Sites which have what are presently thought to be desirable characteristics for subseabed disposal may be found in two of the areas under study.

Radiological predictions were generated for a single repository holding the equivalent of 11 years of high-level waste at the current global production capacity. There would be no releases from correctly

implanted wastes for a few thousand years. The maximum dose which would be experienced by any individual would occur at about 100,000 years after disposal. That dose would be about 10^{-9} Sv per year which is about 1/1,000,000 of average background radiation. The maximum collective dose rate would be about 10^{-2} Sv per year, also at about 100,000 years after disposal. For the present size world population, this is about 1/500,000,000 of background.

The radiological calculations are not very sensitive to uncertainties in the data or in the exact nature of the models. Unless other important processes appear in future validation studies, subseabed disposal should easily exceed standards equivalent to those in use for health and radiological protection today.

MANUSCRIPT NOT AVAILABLE AT TIME OF PUBLICATION

ALBERNI INLET, BRITISH COLUMBIA: A MULTI-USE FJORD SYSTEM WITH FISHERIES RESOURCES AND A LARGE WASTE-DISPOSAL COMPONENT

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Canada V7V 1N6

Alberni Inlet has 5 species of Pacific salmon (Oncorhynchus spp.) and steelhead trout (Salmo gairdneri). A hatchery at Robertson Creek has enhanced coho (O. kisutch) and steelhead production. Great Central and Sproat lakes, where sockeye (O. nerka) rear, have been fertilized to increase productivity. A major pulpmill, producing kraft (sulfate) pulp and newsprint, discharges effluent into the head of Alberni Inlet. Pollution control has been oriented toward protection of the valuable salmonid resources. Dissolved oxygen supply in the upper zone of Alberni Inlet wasshown in early studies to be controlled by the Somass River discharge.

A model was developed relating dissolved oxygen availability in the upper zone to Somass R. discharge. Prediction of the effects of pulpmill effluent was based on the BOD (biochemical oxygen demand) - loading of the system by dissolved organic constituents and on the availability of dissolved oxygen in the upper zone. Toxicity of the effluent was subsequently shown not to be a problem, at least at the acute level, if the dissolved oxygen requirement is met. Somass R. flow was controlled to provide sufficient water and dissolved oxygen during the summer months. Dissolved oxygen concentrations in the harbour waters dropped to dangerously low levels (<5 $mg \cdot L^{-1}$) during the warm, dry summer of 1965. Clarifiers were installed by the pulp mill in 1966 to remove wood solids. In 1970 a biobasin was constructed to reduce the BOD of the effluent and to settle out the fibres. Weekly monitoring of oceanographic variables at a centrally-located station in Alberni Harbour showed improvement in dissolved oxygen levels during the 1970s. Low dissolved oxygen concentrations appeared again in the warm, dry summer of 1985. There has been no clear-cut decline in any of the salmonid species that can be unequivocally attributed to pulpmill effluent disposal. Nevertheless, action may be needed again to ameliorate the depression of dissolved oxygen during dry, warm summers and remove this threat to salmonids.

1. INTRODUCTION

Alberni Inlet is a typical fjord on the southwest coast of Vancouver Island, British Columbia (Figure 1). Because it has numerous tributary streams and lakes at the heads of these streams, Alberni Inlet has always had bountiful salmonid resources. It supports five species of Pacific Salmon (Oncorhynchus spp.) and a popular summer run of steelhead trout (Salmo gairdneri). Sockeye salmon, Oncorhynchus nerka, of the Alberni Inlet system contribute to a valuable commercial fishery, which has been growing over the last few decades in Barkley Sound.

In the terrain surrounding Alberni Inlet are valuable and abundant forest resources. The first sawmill was installed as early as 1861, only a short while after white man entered the scene. The first paper mill was built on the Somass River in 1894. It lasted only two years. A major pulpmill was constructed in 1947, producing 220 tons of unbleached kraft (sulfate) pulp per day. It continues producing to the present day, with its production mainly in newsprint, derived from groundwood supplemented by kraft pulp. Effluent from the initial mill discharged untreated into Alberni Inlet from 1947 to 1966. Various degrees of treatment were instituted as the mill increased its production and modified its product.

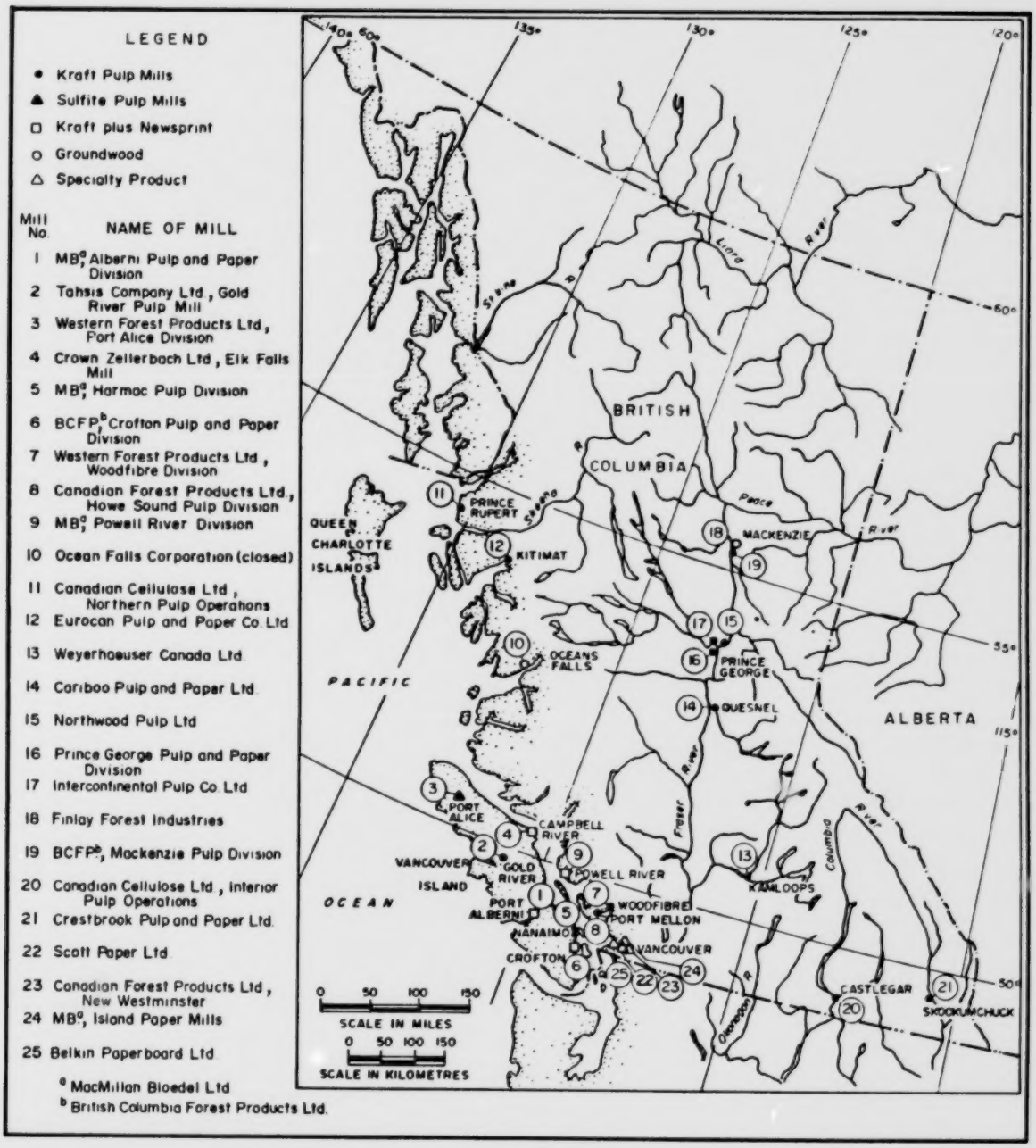
LEGEND
Kraft Pulp Mills
Sulfite Pulp Mills
Kraft plus Newsprint
Groundwood
Specialty Product
Mill No. NAME OF MILL
1 MB[a], Alberni Pulp and Paper Division
2 Tahsis Company Ltd., Gold River Pulp Mill
3 Western Forest Products Ltd., Port Alice Division
4 Crown Zellerbach Ltd., Elk Falls Mill
5 MB[a], Harmac Pulp Division
6 BCFP[b], Crofton Pulp and Paper Division
7 Western Forest Products Ltd., Woodfibre Division
8 Canadian Forest Products Ltd., Howe Sound Pulp Division
9 MB[a], Powell River Division
10 Ocean Falls Corporation (closed)
11 Canadian Cellulose Ltd., Northern Pulp Operations
12 Eurocan Pulp and Paper Co. Ltd.
13 Weyerhaeuser Canada Ltd.
14 Cariboo Pulp and Paper Ltd.
15 Northwood Pulp Ltd.
16 Prince George Pulp and Paper Division
17 Intercontinental Pulp Co. Ltd.
18 Finlay Forest Industries
19 BCFP[b], Mackenzie Pulp Division
20 Canadian Cellulose Ltd., Interior Pulp Operations
21 Crestbrook Pulp and Paper Ltd.
22 Scott Paper Ltd.
23 Canadian Forest Products Ltd., New Westminster
24 MB[a], Island Paper Mills
25 Belkin Paperboard Ltd.
[a] MacMillan Bloedel Ltd.
[b] British Columbia Forest Products Ltd.
BRITISH COLUMBIA
ALBERTA
PACIFIC OCEAN
VANCOUVER ISLAND
QUEEN CHARLOTTE ISLANDS
PRINCE RUPERT
KITIMAT
OCEANS FALLS
PRINCE GEORGE
QUESNEL
MACKENZIE
PORT ALICE
CAMPBELL RIVER
GOLD RIVER
POWELL RIVER
PORT ALBERNI
NANAIMO
WOODFIBRE
PORT MELLON
VANCOUVER
CROFTON
KAMLOOPS
CASTLEGAR
SKOOKUMCHUCK
Liard River
Peace River
Fraser River
Columbia River
Skeena R.
Okanagan R.
140°
135°
130°
125°
120°
60°
55°
50°
115°
0 50 100 150
SCALE IN MILES
0 50 100 150
SCALE IN KILOMETRES

Figure 1.

The purpose of this paper is to briefly review the oceanography of Alberni Inlet, to discuss the mechanisms of assimilation and seaward transport of pulpmill effluent and to review the strategy that has been successful so far in controlling water pollution and preventing a discernible impact on the salmonid resources.

2. OCEANOGRAPHY OF ALBERNI INLET

Tully (1949) conducted the classical work on the oceanography of Alberni Inlet, and related this to the potential for pollution from pulpmill effluent disposal. Alberni Inlet behaves as a typical stratified estuary. It receives most of its freshwater from the Somass River, draining into the head of Alberni Inlet (Figure 2). As the freshwater flows over the surface of the denser seawater, it entrains seawater from below and forms a brackesh upper layer that becomes increasingly more saline as it flows seaward. A longitudinal section through Alberni Inlet is shown somewhat schematically in Figure 3. The upper layer remains comparatively uniform in thickness for the full length of Alberni Inlet.

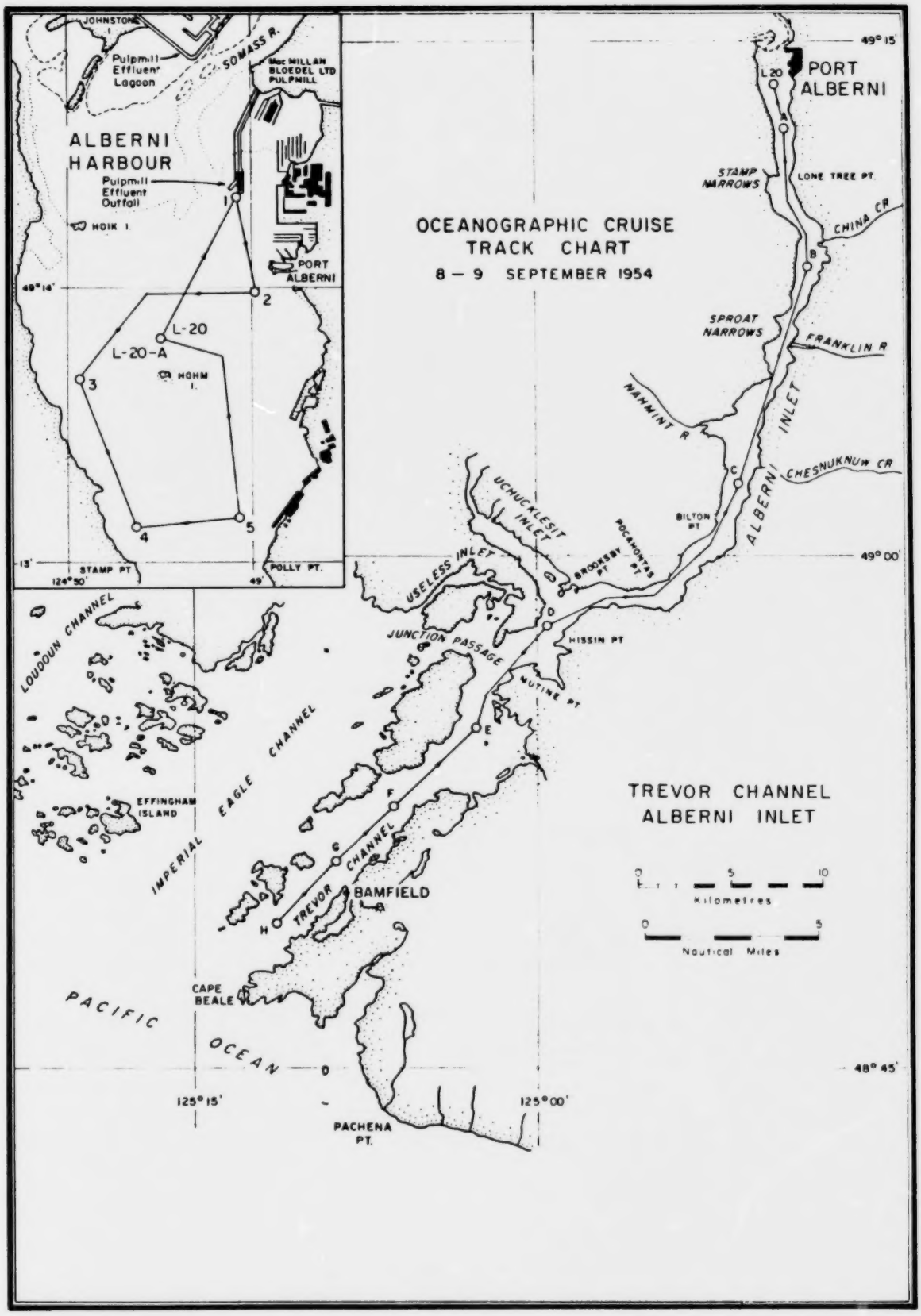
OCEANOGRAPHIC CRUISE
TRACK CHART
8 – 9 SEPTEMBER 1954
ALBERNI HARBOUR
Pulpmill Effluent Lagoon
JOHNSTONE I.
SOMASS R.
MacMILLAN BLOEDEL LTD PULPMILL
Pulpmill Effluent Outfall
HOIK I.
PORT ALBERNI
L-20
L-20-A
HOHM I.
STAMP PT.
POLLY PT.
124°30'
49°
49°14'
49°13'
PORT ALBERNI
L20
STAMP NARROWS
LONE TREE PT.
CHINA CR.
SPROAT NARROWS
FRANKLIN R.
NAHMINT R.
ALBERNI INLET
CHESNUKNUW CR.
UCHUCKLESIT INLET
BILTON PT.
POCAHONTAS PT.
BROOKSBY PT.
USELESS INLET
JUNCTION PASSAGE
HISSIN PT.
MUTINE PT.
LOUDOUN CHANNEL
IMPERIAL EAGLE CHANNEL
EFFINGHAM ISLAND
TREVOR CHANNEL
BAMFIELD
CAPE BEALE
PACIFIC OCEAN
PACHENA PT.
125°15'
125°00'
49°00'
48°45'
TREVOR CHANNEL
ALBERNI INLET
Kilometres
Nautical Miles

Figure 2

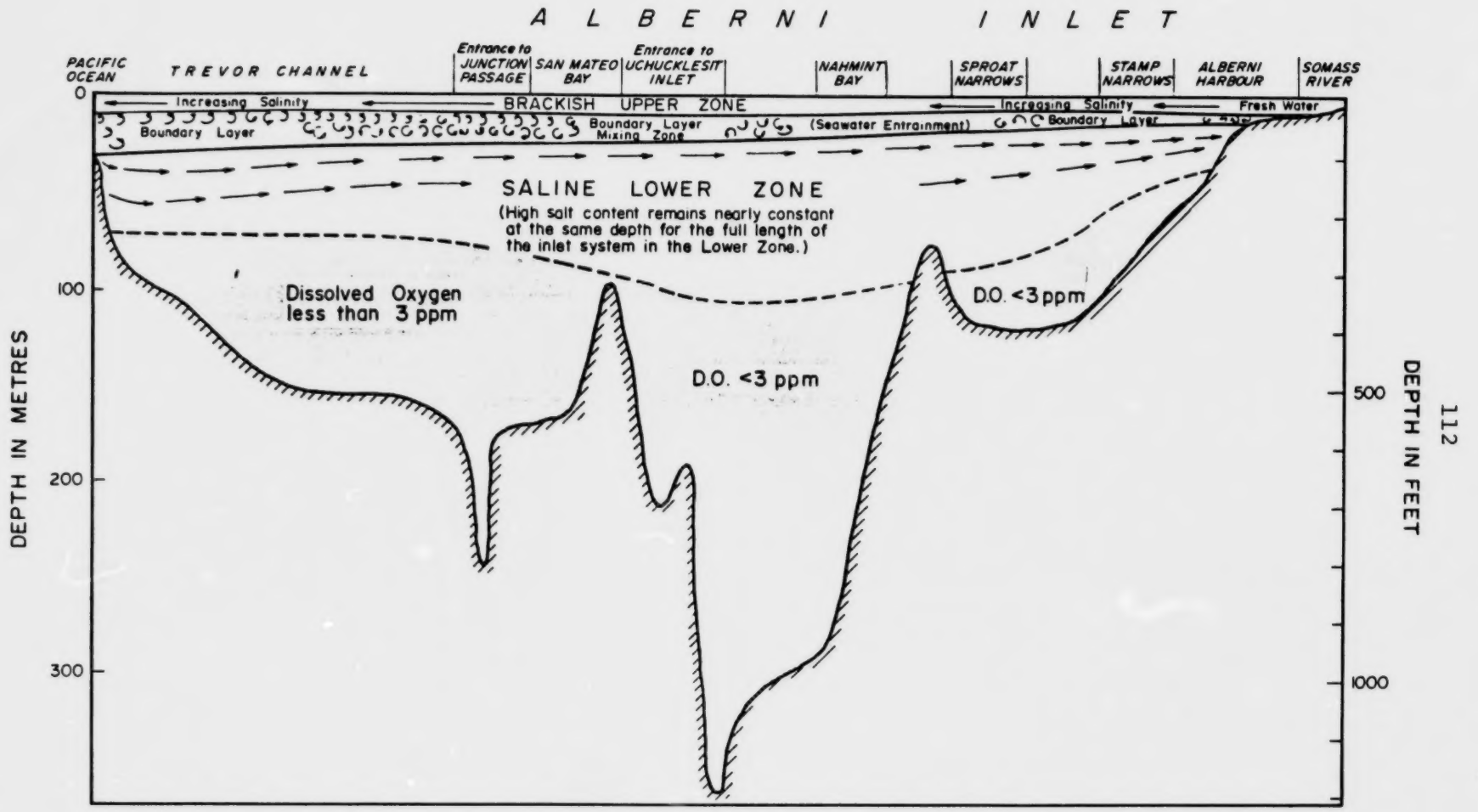
A L B E R N I I N L E T
PACIFIC OCEAN
TREVOR CHANNEL
Entrance to JUNCTION PASSAGE
SAN MATEO BAY
Entrance to UCHUCKLESIT INLET
NAHMINT BAY
SPROAT NARROWS
STAMP NARROWS
ALBERNI HARBOUR
SOMASS RIVER
0
Increasing Salinity
BRACKISH UPPER ZONE
Increasing Salinity
Fresh Water
Boundary Layer
Boundary Layer Mixing Zone
(Seawater Entrainment)
Boundary Layer
SALINE LOWER ZONE
(High salt content remains nearly constant at the same depth for the full length of the inlet system in the Lower Zone.)
Dissolved Oxygen less than 3 ppm
D.O. <3 ppm
D.O. <3 ppm
100
200
300
DEPTH IN METRES
500
1000
DEPTH IN FEET

Figure 3

Because of an increasing amount of seawater in the layer as it flows seaward, the net seaward current velocity increases. In order to replace the seawater lost by entrainment to the surface layer, there is a net inflow of seawater below the upper layer. Tully (1949) recommended that any effluent discharged into Alberni Inlet should be kept in the upper zone so that advantage could be taken of the rapid seaward transport in that zone.

Because pulpmill effluent has a certain BOD (biochemical oxygen demand), it can have an impact on the dissolved oxygen concentration in the receiving water. Tully's (1949) research stressed the importance of dissolved oxygen in the Alberni Inlet system to protect the salmonid resource. He investigated a number of variables that would influence the amount of dissolved oxygen in the upper zone. Among these were the wind effect in aeration of the surface waters and the effect of the Somass River discharge. The seasonal variation of the Somass River discharge is shown in Figure 4. Tully (1949) developed a relationship between the Somass River discharge and the amount of dissolved oxygen present in the upper zone of Alberni Inlet. This relationship has been used to control pulpmill pollution, indeed pulpmill production, at the Port Alberni pulpmill during the 1950's and 1960's (Waldichuk, 1955).

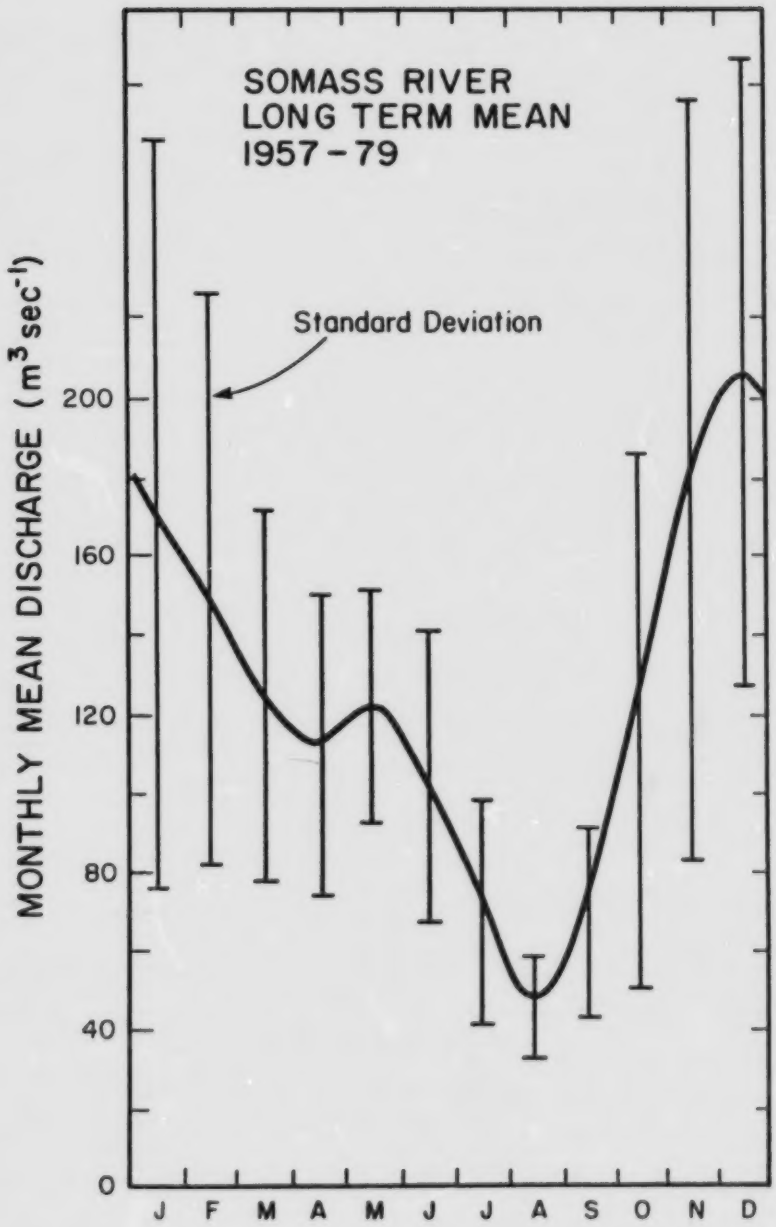
SOMASS RIVER
LONG TERM MEAN
1957 - 79
Standard Deviation
MONTHLY MEAN DISCHARGE ($m^3 sec^{-1}$)
0
40
80
120
160
200
J F M A M J J A S O N D

Figure 4

3. EFFECT OF PULPMILL EFFLUENT IN ALBERNI INLET

The initial proposal of the forest company planning a pulpmill in Port Alberni during the 1930's was to install a sulfite mill. Effluent from such a mill was often notoriously high in BOD, because no recovery was practised. It was shown by Tully (1949) that a sulfite mill with a nominal production of 300 tons per day, discharging its effluent into Alberni Inlet, would have to close for part of the year, because of the high BOD loading and the severe depression in dissolved oxygen it would incur during the summer months.

Fortunately, because of changing market demands, the forest company decided by the mid-1940s to install a kraft pulpmill instead of sulfite. The kraft process involves recovery of the "black liquor", following digestion of wood chips at elevated temperature and pressure. The black liquor is evaporated and burned for energy as well as for the salts it contains. Effluent from a kraft mill typically contains only wash waters of assorted types, except if bleaching is practised, in which case bleach wastes are discharged as well. The BOD of kraftmill effluent is only about 10% of unrecovered sulfite-mill effluent.

The initial unbleached kraft pulpmill of 220 tons per day in Port Alberni was considered inadequate to meet market demands by the mid 1950's. Management desired to expand into full-bleached kraft production at about 500 tons per day. There was a need then to conduct BOD and toxicity tests on effluent from such a mill. Studies showed that effluent from the Harmac pulpmill near Nanaimo, which is a full-bleach kraft mill, had a BOD of about 100 $mg.L^{-1}$ (Greer *et al.*, 1956). Bioassays conducted on the same effluent showed that salmonids could withstand a concentration of about 5% of the effluent over a 4-day period without significant mortality, (Alderdice and Brett, 1957). Analyses of the two sets of data showed that if enough water were provided by the Somass River to satisfy the dissolved oxygen requirement, then toxicity would not arise as a problem. The flow of water required from the Somass River to provide the dissolved oxygen to meet the needs for stabilization of effluent from a 500 ton per day full-bleach kraft mill and leave adequate oxygen for salmonids is shown in Figure 5.

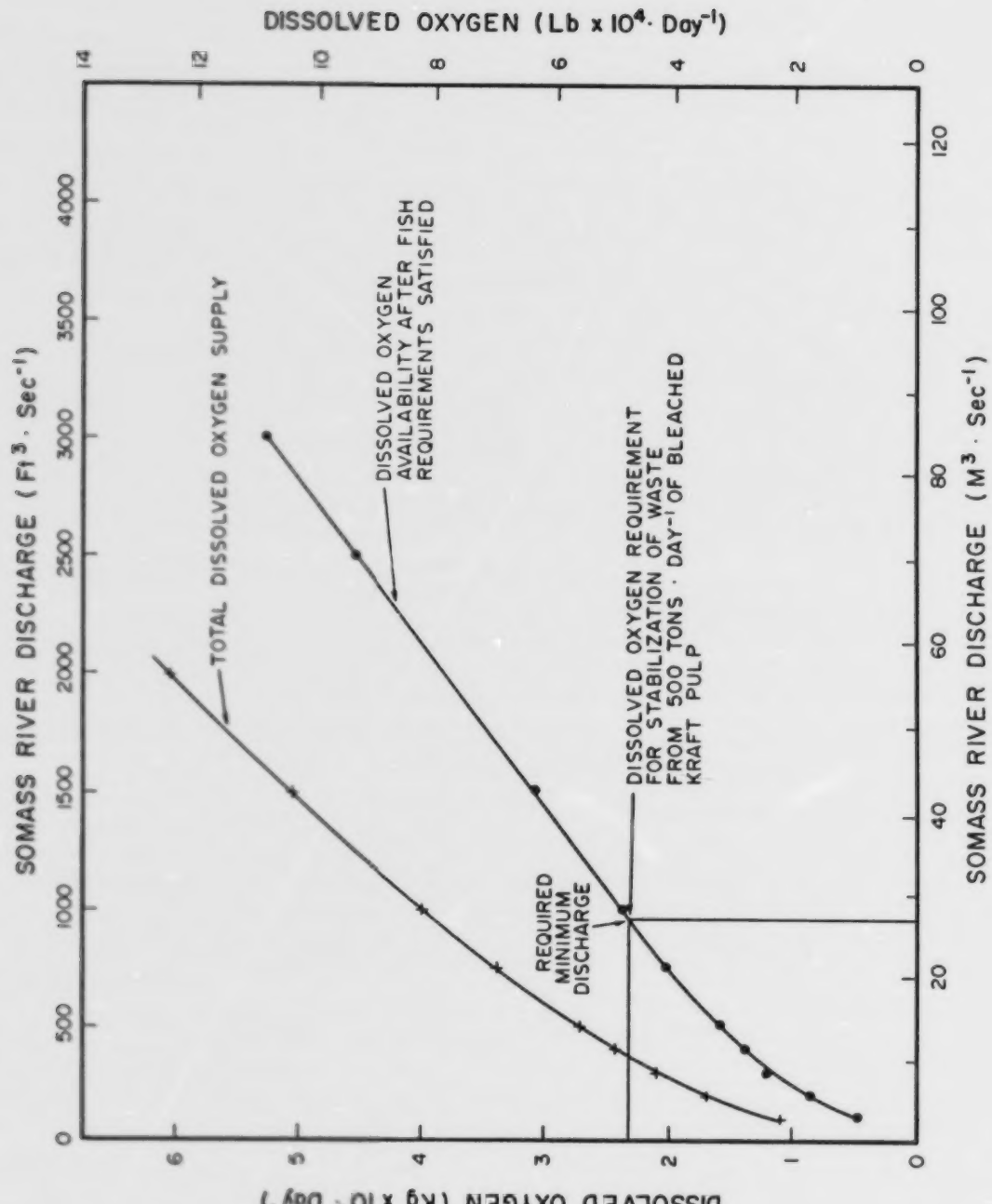
DISSOLVED OXYGEN (Lb x 10⁴ · Day⁻¹)
SOMASS RIVER DISCHARGE (Ft³ · Sec⁻¹)
TOTAL DISSOLVED OXYGEN SUPPLY
DISSOLVED OXYGEN AVAILABILITY AFTER FISH REQUIREMENTS SATISFIED
DISSOLVED OXYGEN REQUIREMENT FOR STABILIZATION OF WASTE FROM 500 TONS · DAY⁻¹ OF BLEACHED KRAFT PULP
REQUIRED MINIMUM DISCHARGE
DISSOLVED OXYGEN (Kg x 10⁴ · Day⁻¹)
SOMASS RIVER DISCHARGE (M³ · Sec⁻¹)

Figure 5

The pulpmill at Port Alberni did not expand ultimately into a 500-ton per day full-bleach kraft operation, but instead expanded in groundwood production for newsprint. Nevertheless, the discharge of the Somass River, through a dam on one of its tributaries, the Stamp River, was controlled to meet the dissolved oxygen requirements in Alberni Inlet, based on the BOD of the total pulpmill effluent. This continued to be the technique for pulpmill pollution control in Alberni Inlet for many years.

4. OCEANOGRAPHIC MONITORING OF ALBERNI INLET

Alberni Inlet was regularly surveyed oceanographically during the 1950's and 1960's (Waldichuk 1954, 1956, 1959, 1962; Waldichuk et al., 1968). It was recognized by both the pulpmill company in Port Alberni and by the Department of Fisheries that it would be desirable to have a station near the middle of Alberni Harbour to monitor conditions weekly. Such a station was established near Hohm Is. (see Figure 2), and sampling of the water at various depths was commenced in 1958 (Waldichuk et al., 1969). Dissolved oxygen was determined as well as salinity and temperature. The program continues to the present. The seasonal variations of dissolved oxygen at various depths at this station for the first year of weekly monitoring are shown in Figure 6. It is obvious that there is a large decline in dissolved oxygen, especially near the bottom, during the summer.

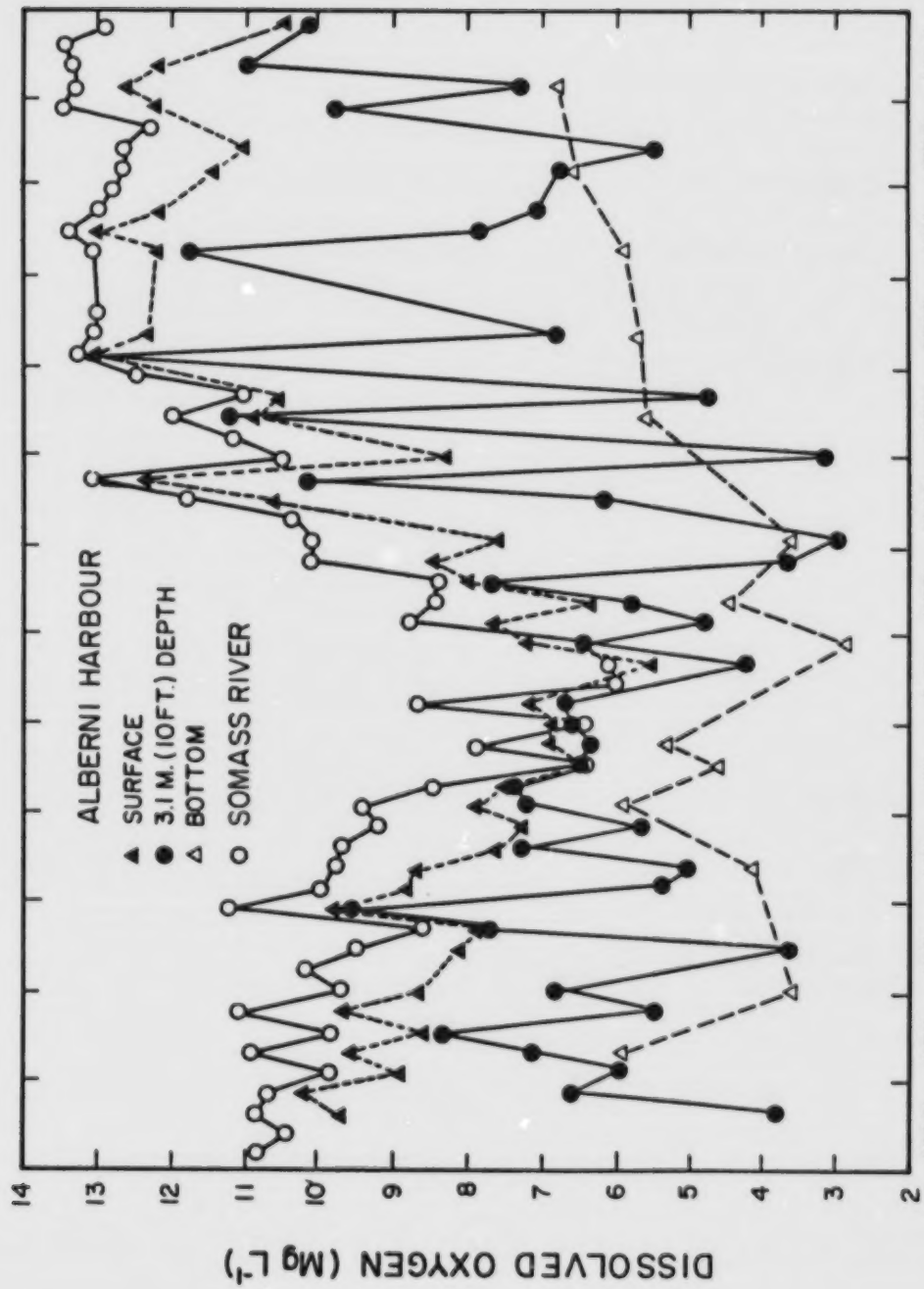
ALBERNI HARBOUR
▲ SURFACE
● 3.1 M. (10 FT.) DEPTH
△ BOTTOM
○ SOMASS RIVER
DISSOLVED OXYGEN (Mg L⁻¹)
14
13
12
11
10
9
8
7
6
5
4
3
2
MAR APR MAY JUN JUL AUG SEP OCT NOV DEC JAN FEB MAR
1958
1959

Figure 6

It was during monthly surveys of the upper end of Alberni Inlet in 1965-66 (Waldichuk et al., 1968) that we noted alarmingly low dissolved oxygen concentrations in Alberni Harbour. This was brought to the attention of authorities in the Department of Fisheries (Waldichuk, unpublished), and action was soon undertaken by the company (Stanley Associates Engineering Ltd., 1966). It was decided that one way improvements could be made in the dissolved oxygen concentration of Alberni Harbour was to remove the wood solids from the woodroom effluent by installation of clarifiers. This was done in 1966. Subsequently in 1970, a biobasin was also installed to biologically treat the effluent and thus reduce the BOD.

Profiles of dissolved oxygen for different years, starting in 1941 before the pulpmill was constructed (Pacific Oceanographic Group, 1957), are shown in Figure 7. The oxygen profile for 16 August 1965 is in considerable contrast with profiles of other years shown to the right. It is noteworthy that at depths greater than 2m the oxygen profile for 16 August 1985 exhibits dissolved oxygen concentrations substantially lower than that for 16 August 1965. It suggests that migrating salmonids had a relatively thin surface layer through which to swim without suffering

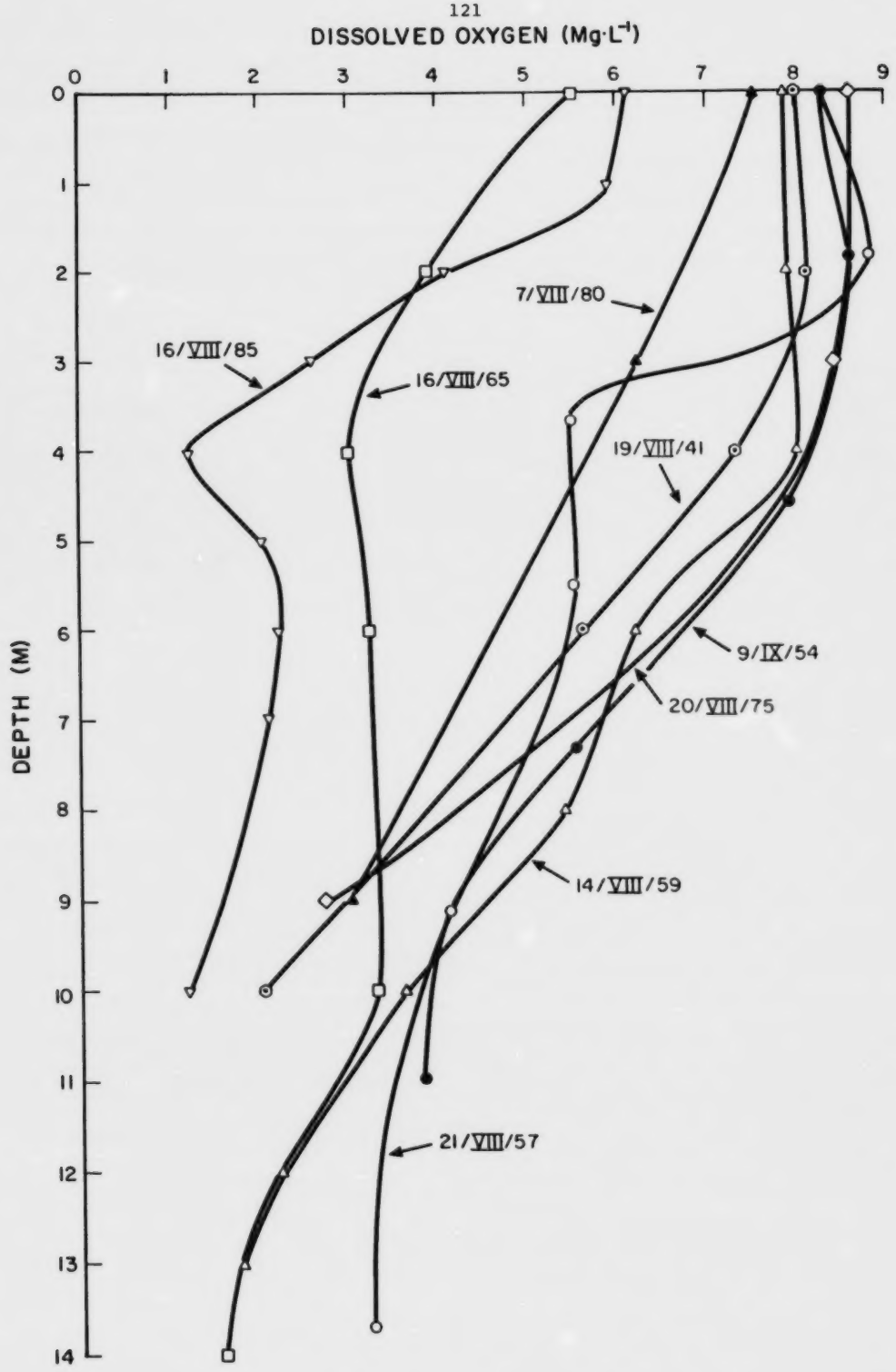
DISSOLVED OXYGEN (Mg·L⁻¹)
0
1
2
3
4
5
6
7
8
9
DEPTH (M)
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
7/VIII/80
16/VIII/85
16/VIII/65
19/VIII/41
9/IX/54
20/VIII/75
14/VIII/59
21/VIII/57

Figure 7

low-oxygen stress. The remainder of the water column is considerably below Level B (6.5 mg.L^{-1} at 20°C), recommended as a minimum for salmonids (Davis, 1975). It may even be regarded as Level C,under which "it is possible that a large portion of a given fish population or fish community may be severely affected by low oxygen, especially on a 'chronic exposure' basis" (Davis, 1975).

5. FISHERIES IN THE ALBERNI INLET SYSTEM

Alberni Inlet is well endowed with fisheries resources, particularly salmonids. It competes with another famous salmon fishing area on Vancouver Island, Campbell River, for the title "Salmon Capital of the World". All five species of Pacific salmon , genus Oncorhynchus, pass through Alberni Inlet to and from its tributary streams and lakes. Alberni Inlet is famous for its huge chinooks, Oncorhynchus tshawytscha, locally referred to as "tyee", that enter its waters in late summer and are considered to match the size of Campbell River tyee. The Somass River and its tributaries, the Stamp and Sproat rivers, are well known to anglers for the summer run of steelhead, Salmo gairdneri, which is now being enhanced by a hatchery at Robertson Creek.

The major salmon resource in the Alberni Inlet system, however, is the sockeye salmon, Oncorhynchus nerka. It supports an important commercial fishery in Barkley Sound.

There are basically three stocks of sockeye in Barkley Sound, which are indigenous to three lakes draining into the Sound: (1) Great Central Lake; (2) Sproat Lake; and (3) Henderson Lake (see Figure 8). The first two lakes drain into the head of Alberni Inlet by way of the Somass River. Salmon migrating to and from these lakes could be directly affected by effluent from the pulpmill at Port Alberni. Henderson Lake drains into the lower end of Alberni Inlet. Its salmon population is less likely to be affected by the pulpmill effluent. The three stocks of sockeye in Barkley Sound, originating from the three lake systems, can be identified quite accurately by brain parasites (L. Margolis, personal communication). This has allowed close management of the different stocks of sockeye in recent years. The commercial fishery has been tightly controlled to allow adequate escapement for the successful perpetuation of the three stocks. Fertilization of Great Central Lake commenced in 1970 to increase primary productivity and ultimately sockeye production (LeBrasseur et al., 1978). Fertilization of the other two lakes has been conducted subsequently (Hyatt and Stockner, 1985).

The changes in Barkley Sound sockeye production over the last century have been chronicled by Hyatt and Steer (1987) and summarized by Hyatt (1986). Since 1950, which coincides with the period of pulpmill production at Port Alberni, there has been a steady increase in sockeye production, which was

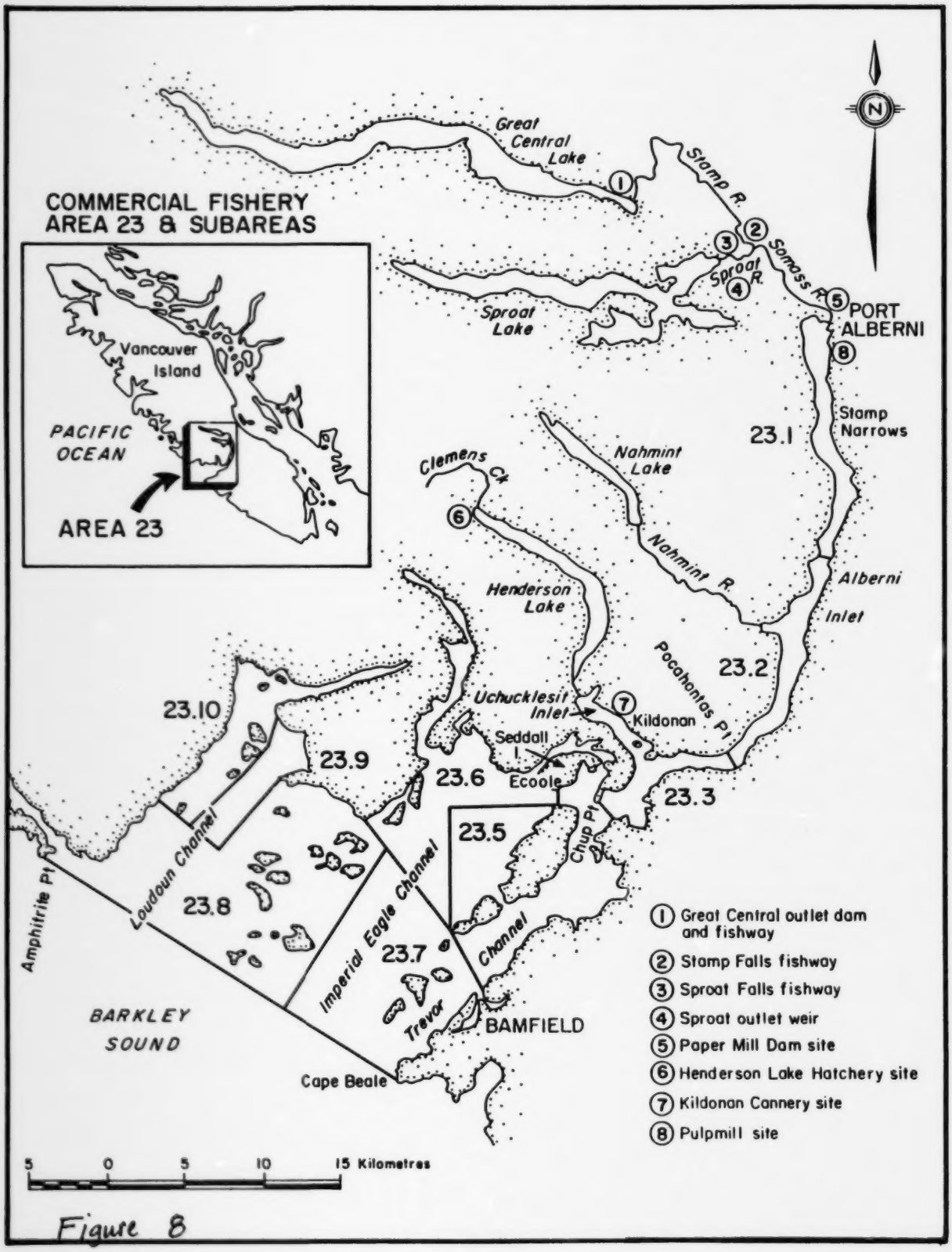
COMMERCIAL FISHERY
AREA 23 & SUBAREAS
Vancouver Island
PACIFIC OCEAN
AREA 23
Great Central Lake
Stamp R.
Somass R.
Sproat R.
Sproat Lake
PORT ALBERNI
Stamp Narrows
23.1
Clemens Cr.
Nahmint Lake
Nahmint R.
Alberni Inlet
Henderson Lake
Pocahontas Pt.
23.2
Uchucklesit Inlet
Kildonan
Seddall I.
Ecoole
23.10
23.9
23.6
23.3
23.5
Chup Pt.
Loudoun Channel
Amphitrite Pt.
23.8
Imperial Eagle Channel
23.7
Trevor Channel
BARKLEY SOUND
BAMFIELD
Cape Beale
5 0 5 10 15 Kilometres
(1) Great Central outlet dam and fishway
(2) Stamp Falls fishway
(3) Sproat Falls fishway
(4) Sproat outlet weir
(5) Paper Mill Dam site
(6) Henderson Lake Hatchery site
(7) Kildonan Cannery site
(8) Pulpmill site

Figure 8

quite dramatic after 1970 (Figure 9). This has continued to the present time, except for 1985 and 1986. The drop in sockeye returns to a level below expectations during those two years was attributed to poor ocean survival. The returns during 1987 were above expectations, at about 650,000, although the sockeye returned rather later than usual.

6. DISCUSSION

Alberni Inlet can be truly considered a multiple use area, where the different uses could be in serious conflict. The various uses and their impacts on the Somass River estuary and on the fisheries resources of the Alberni Inlet system have been reviewed by Morris and Leaney (1980). There is no question that the use of Alberni Inlet by the forest industry, from log storage to pulpmill effluent discharge, has had an impact on the estuarine ecosystem. The deposition of wood solids in Alberni Harbour has led to anoxic conditions in the sediments and fish kills during dredging (Hourston and Herlinveaux, 1957).

Dredging is an annual event in Alberni Harbour, because of its shallow depth and the need to maintain navigational channels for deep-sea ships. The dredged material is disposed of at a designated dumpsite in Alberni Inlet, south of Alberni Harbour, between Polly Point and Stamp Narrows (see Figure 2). The biological effects of the wood-rich dredged material have been investigated by Levings et al. (1985). They showed changes in biological communities with

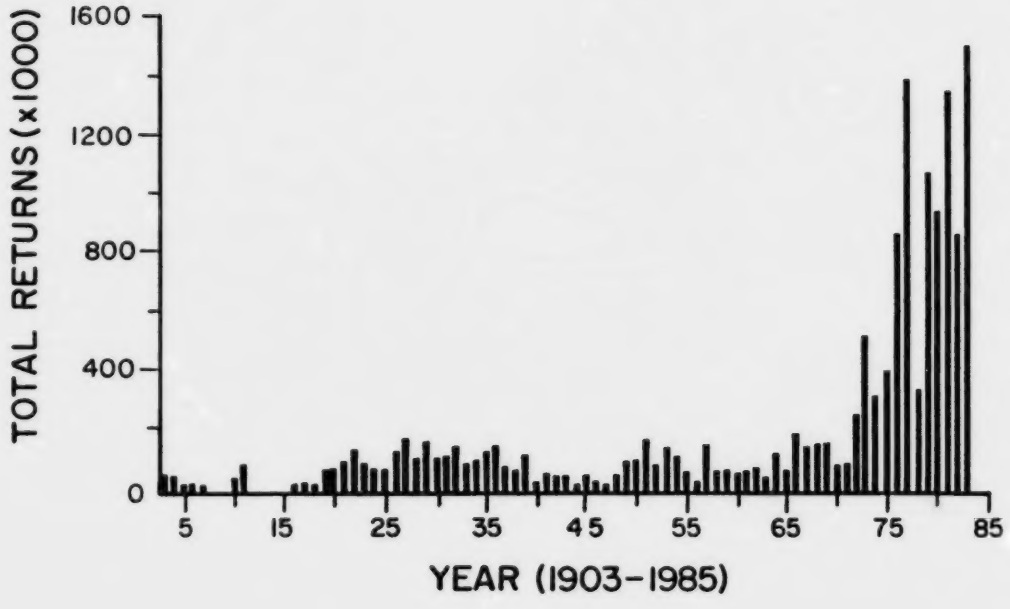
TOTAL RETURNS (x1000)
1600
1200
800
400
0
5
15
25
35
45
55
65
75
85
YEAR (1903-1985)

Figure 9.

an increase in opportunistic species, such as Capitella spp., and a decrease in certain sensitive species. For example, the burrowing shrimp, Axiopsis spinulicauda, occurred in 25% of the bottom grabs for the control site compared to 6% of grabs at the dumpsite. There is little evidence of life either in or on the sediments of Alberni Harbour, judging from observations in a bottom-sampling program conducted in July 1987 (D. Brothers, personal communication).

Dissolved oxygen in the water column of Alberni Harbour continues to be the critical variable in the ecosystem of the harbour, as it pertains to the effect of the effluent from the Port Alberni pulpmill on the salmonid resources. The dissolved organic constituents in the effluent probably still have the major impact on the dissolved oxygen concentration in the water in spite of the biological treatment. The reduction of dissolved oxygen supply by photosynthesis through the retardation of light penetration by the coloured constituents in the effluent has also been proposed as an important factor in the dissolved oxygen budget (Parker and Sibert, 1973; Sibert and Parker, 1973). It has been estimated that 10% of the dissolved oxygen depletion in the water column of Alberni Harbour is due to decomposition of wood solids deposited on the bottom (Stanley Associates Engineering Ltd., 1966).

The weekly monitoring program at the Hohm Island station (Waldichuk et al., 1969) provides a useful index and an early

warning on depressed dissolved oxygen concentrations in Alberni Harbour. Data from this program and from monthly surveys in Alberni Inlet during 1965-1966 (Waldichuk et al., 1968) formed the basis for alerting government authorities, as well as pulpmill management of dangerous trends in dissolved oxygen concentrations in Alberni Harbour (Waldichuk, unpublished report). An improvement was effected in dissolved oxygen concentrations following installation of clarifiers in 1966 and a biobasin in 1970.

A decline in dissolved oxygen of the water column of Alberni Harbour was noted again during August 1985. This was a period of warm, dry weather. Similar weather conditions occurred in the summers of 1986 and 1987. The oceanographic data from the Hohm Island station for those years have not yet been examined. It is expected, however, that they were similar to those of 1985. Unfortunately, adult sockeye normally migrate through Alberni Harbour during the period when dissolved oxygen concentrations are expected to be at their lowest and temperatures at their highest. The 1985 dissolved oxygen data suggest that they can only migrate through the upper 2-m layer without severe low-oxygen impact. Even in this layer the dissolved oxygen concentration barely met the B Level (6.5 $mg \cdot L^{-1}$) of protection at 20° C recommended by Davis (1975). B Level is considered to result in moderate risk to aquatic organisms in that some species may develop symptoms of oxygen stress. Whether migrating juvenile and adult salmon experience stress from depressed

dissolved oxygen levels is uncertain. However, dissolved oxygen concentrations appear to have reached such a low level in late summer, and the layer of water through which salmon can safely move has become so shallow, that an accidental spill of oil or other toxic substance that drives the fish down could prove disastrous to a salmon run.

The long-term trends in sockeye production for Barkley Sound do not suggest a significant impact from the pulpmill so far. So many factors are involved in the size of sockeye stock that any minor impact from pulpmill effluent would be obscured. It is noteworthy, for example, that the proportion of the Henderson Lake stock to the Barkley Sound sockeye fishery has declined in recent years in relation to the Great Central Lake and Sproat Lake stocks. The Henderson Lake stock would be the least affected by pulpmill effluent of the three stocks. Clearly, other factors, such as careful management of the fishery, have been more important in determining the size of sockeye stocks than the existing level of pulpmill pollution. Installation of fishways in 1927 and 1950 for the sockeye to get around Stamp Falls and Sproat Falls, respectively, have been important elements in building up the sockeye stocks of Great Central and Sproat lakes. Ocean survival of the different stocks is still uncertain, although it is believed that years of warm water, such as those of 1982-83 associated with El Niño, can be disastrous for salmon survival. Lake enrichment may have contributed to larger than expected returns of sockeye during certain years.

Perhaps some luck has been involved, as well as good pollution control, in that no major disaster has occurred with respect to mass destruction of salmon by low dissolved oxygen due to pulpmill pollution in Alberni Harbour. The salmon must move through this harbour in order to reach their spawning grounds in streams tributary to the Somass River and to Great Central and Sproat lakes. A low-dissolved oxygen barrier could prevent them from reaching their destination. Fortunately, this has not happened so far, but from the trends shown in Figure 6 it is difficult to have confidence that the successful passage of salmon through Alberni Harbour will endure without some corrective action.

7. CONCLUSIONS

(1) Dissolved oxygen (DO) continues to be a reasonable measure of aquatic environmental quality in Alberni Harbour.

(2) Lethal conditions for salmonids have not arisen in Alberni Harbour since the pulpmill was installed in 1947. Any sublethal effects on salmonids from depressed DO levels and the presence of contaminants are unknown.

(3) Following the dangerously low DO concentrations in Alberni Harbour during August 1965, improvements in summer DO levels coincided with installation of clarifiers to remove wood solids in 1966 and a biobasin for biological treatment of the effluent in 1970.

(4) Low DO levels occurred again in Alberni Harbour below 2 m depth during the warm, dry summer of 1985. DO concentrations dropped to less that 3 mg. L^{-1} , which is well below the 7.5 mg.L^{-1} considered to be optimum for anadromous species at 20°C.

(5) Sockeye salmon (Oncorhynchus nerka) stocks in the Alberni Inlet system have continued to increase during the period 1960-1983. This must be credited largely to improved salmon management, allowing larger escapement to the spawning grounds. Other factors may have been responsible, in part, for this increase.

(6) Low returns, below expectations, of sockeye salmon during 1985 and 1986 have been attributed to poor ocean survival.

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LIST OF FIGURES

Figure 1. Locations of pulp and paper mills in British Columbia.

Figure 2. Chart of Alberni Inlet, showing locations of stations occupied during oceanographic surveys.

Figure 3. Diagrammatic longitudinal section through Alberni Inlet, showing the different processes, water masses and dissolved oxygen levels. (From Waldichuk, 1962).

Figure 4. Seasonal variation of Somass River discharge. (From Hyatt and Steer, 1987).

Figure 5. Relationship of the dissolved oxygen supply in the upper zone of Alberni Inlet to Somass River discharge.

Figure 6. Seasonal changes in dissolved oxygen at various depths at the Hohm Island station in Alberni Harbour. (From Waldichuk, 1959).

Figure 7. Profiles of dissolved oxygen concentration at the Hohm Island station during the late summer of different years.

Figure 8. Chart showing the various lakes and streams contributing sockeye to Barkley Sound. (From Hyatt and Steer, 1987).

Figure 9. Year-to-year variation of total returns of sockeye to Barkley Sound between 1903 and 1983. (From Hyatt, 1986).

FIGURE CAPTIONS

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ENVIRONMENTAL IMPACT OF SOLID WASTE DISPOSAL ON THE COASTAL AND MARINE ECOSYSTEMS OF KENYA

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The Coastal and Off-shore Ecosystems of Kenya contain natural resources that are of considerable significance to the economic and social development of the country. These resources are however threatened by adverse effects of pollution. Marine pollution along the Kenya Coast originates principally from land-based sources, wastes and maritime activities.

Major sources of pollution on the Kenyan coastal and marine environment with adverse effects are sewage, domestic and commercial wastes discharges, industrial waste outfalls, agricultural waste disposal, river run-off and riverine transfer of pollutants from land, mining activities, maritime and oil tanker traffic, exploration of sea-bed resources, coral mining, coastal construction, dredging and ocean dumping.

Major sources of solid wastes are dumping of domestic and industrial wastes on tipping sites. Solid wastes dumped at various Municipal landfills along the Kenya coast consist of broken glass material, metal containers, paper, cardboard materials, hazardous wastes, lubricating oils in containers, expired medicine, old toxic chemicals and animal

cadavers. Another source of solid wastes is disposal from foreign ships as well as disposal of unclaimed cargo from the Mombasa harbour. Aesthetic aspects of marine pollution includes littering of beaches with bottles, plastics, metal cans, old tyres, etc. No precautionary measures are taken to prevent landfill leakages from polluting the marine environment nor to circumvent run-off of leachates during the rainy season. This mode of solid waste management is hazardous because the leachates are high in BOD and contain large amounts of dissolved toxic metals and persistent toxic organic chemicals which have deleterious effects on flora and fauna of the coastal and marine ecosystems of Kenya.

Under existing conditions of increasing human population and development activities along the Kenya coast, alternative solid waste management methods/techniques are not only desirable but a must if Kenya has to meet its coastal developmental goals and marine environment conservation objectives.

MANUSCRIPT NOT AVAILABLE AT TIME OF PUBLICATION

A New Technique for the Rapid Identification of Sewage and Sludge In the Marine Environment Using Light Microscopy

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ABSTRACT

The newly developed Rapid Sewage/Sludge Indicator Technique (RSIT) of detecting sewage contamination in ocean water and bottom sediments is described. It is a rapid indicator for use during monitoring, and especially for sewage-related emergencies, to determine where and to what extent marine environments are contaminated.

The new method utilizes microscopic examination of samples for the xylem tracheary elements of terrestrial plant vascular tissue as identifiers of sewage and sludge. This material, found in the roughage of the human diet, passes through the digestive system intact and can be identified easily in samples. As these vascular tissues are not normally found in ocean waters, they are an indicator of human fecal contamination. Quantitative procedures take 45 minutes to an hour, but for gross screening purposes, determinations can be made within five minutes.

Results of a recent study under the auspices of the National Oceanic and Atmospheric Administration (NOAA) demonstrated the validity of the RSIT. The technique has proven useful for resource management and for situations in which immediate information is necessary for public health decisions. Examples from Nassau County include directing estuarine and coastal cleanup, assessing extent of sewage sludge movement from offshore dump sites, and determining the need for beach closures following accidental input.

BACKGROUND

The use of the marine environment as a repository of sewage-related material has necessitated the development of monitoring techniques in an attempt to determine location and effects. Some tracers in current use include heavy metals, Clostridium perfringens spores, fecal steroids (coprostanol), total organic carbon, and the ratio of carbohydrates to total organic carbon (Charnell, 1975; Hatcher and McGillivary, 1979; Hatcher and Keister, 1976). Many of these techniques, while accurately assessing the effects and paths of the raw sewage or sewage sludge, rarely yield immediate results, are highly sophisticated and time-consuming, and require expensive equipment which must be operated by highly trained technicians. A need exists during monitoring, and expecially in times of sewage-related emergency, for an indicator that can be used rapidly and on- site to determine if, and to what extent, areas of the marine environment are contaminated. Another matter of great importance is to judge if a correct determination has been reached between perceived/assumed and actual sewage or sludge when the observation has been made by a citizen filing a complaint.

The Rapid Sewage/Sludge Indicator Technique (RSIT) was developed during the early 1970's as a response to critical need during the performance of duties of the Marine Biologist in the Bureau of Water Pollution Control for the Nassau County Department of Health.

Nassau County is the western territory of Long Island, just to the east of New York City. The southern barrier beach ocean shoreline forms one boundary of the New York Bight, where disposal of sewage sludge has occurred 12 miles offshore for 60 years since 1924. This is the nation's largest ocean dumping operation, serving almost 20 million people. In 1980, for example, the dump site received 7.2 million tons of sewage sludge.

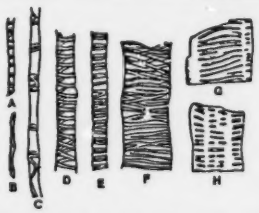
In Nassau County, the RSIT was used during sewage-related crisis situations to:

- direct cleanup efforts after a massive sludge storage tank explosion (Freudenthal, 1976) and assorted sludge pipe breaks sent sludge onto tidal flats, wetlands and salt marsh islands, and into the open bay
- determine which beaches should be closed when untreated raw sewage from electrically incapacitated New York City sewage treatment plants flowed into Nassau County marine waters after "black outs" during the summer swimming season
- ascertain if the leading edge of sludge from the 12-mile dump site was advancing toward the beaches (Nassau County Department of Health, 1976)
- judge if numerous citizen complaints of sewage/sludge on beaches or in the marine environment were valid.

RSIT CONCEPTS

- The vascular system of food plants is composed of xylem, the principal water conducting-tissue, and phloem, the food-conducting tissue. The xylem also functions as the mechanical support mechanism of the plant. The cells of the xylem have enduring rigid walls composed primarily of cellulose which is further strengthened by a secondary lignified wall. The lignin is deposited on the inner surface of the cell wall, replacing most of the cellulose. On the basis of the manner in which lignin becomes deposited, the secondary walls may appear as spirals, rings, parallel bars, irregular networks or pits (Figure 1), (Bold 1977, Esau 1965).

Figure 1: SECONDARY WALL STRUCTURE OF TRACHEARY ELEMENTS



A. Annular
B. Annular extended
C. Annular, transitional to helical
D. Helical
E. Helical
F. Reticulate
G. Scalariformly pitted
H. Oppositely pitted

From: K. Esau, Plant Anatomy (2nd Ed.), John Wiley and Sons, Inc. N.Y., p. 233

- The xylem tissue, composed of cellulose, hemicellulose, and lignin is the most important component in dietary fiber, that fraction of food resistant to attack by the mammalian digestive enzymes. Most fruits and vegetables are included in fiber-contributing food groups. Examples of food that have high fiber content are carrots, lettuce (Figure 2), cabbage (Figure 3), celery and fennel. Fiber is minimally degraded in the digestive process. Cellulose may be partly degraded by the colon microflora of man, but

lignins are very resistant to colonic degradation and chemical attack. The human body passes the undigested remains of the food fiber out of the body as the roughage component of fecal material (Davenport, 1977; Inglett, 1979).

Figure 2: LETTUCE XYLEM TISSUE

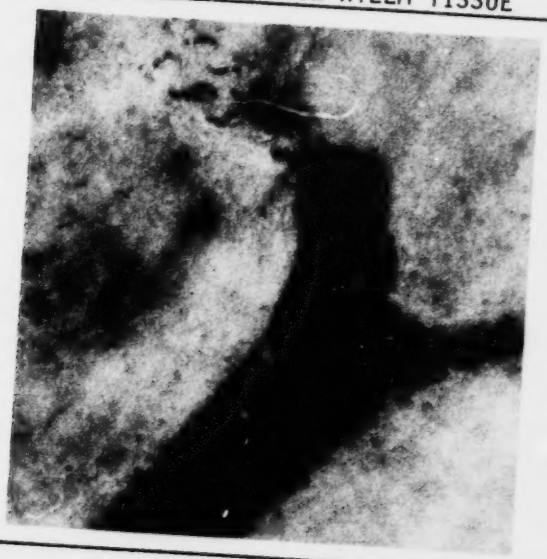


Figure 3: RED CABBAGE XYLEM TISSUE

- Sewage treatment may alter, but does not destroy (Figure 4), the identifiable fiber remains present in raw sewage and sludge. Primary sewage treatment consists of mechanical separation and settling of solids from wastewater, while the secondary stage consists of biological treatment of the separated components of the raw sewage. The solids removed from wastewater in both primary and secondary treatment units, together with the water removed with them, constitute wastewater sludge (NYSDEC, 1984). The fiber settles during primary treatment and remains as part of the sludge, through secondary treatment, until disposal (Freudenthal, 1984).

Figure 4: XYLEM IDENTIFICATION

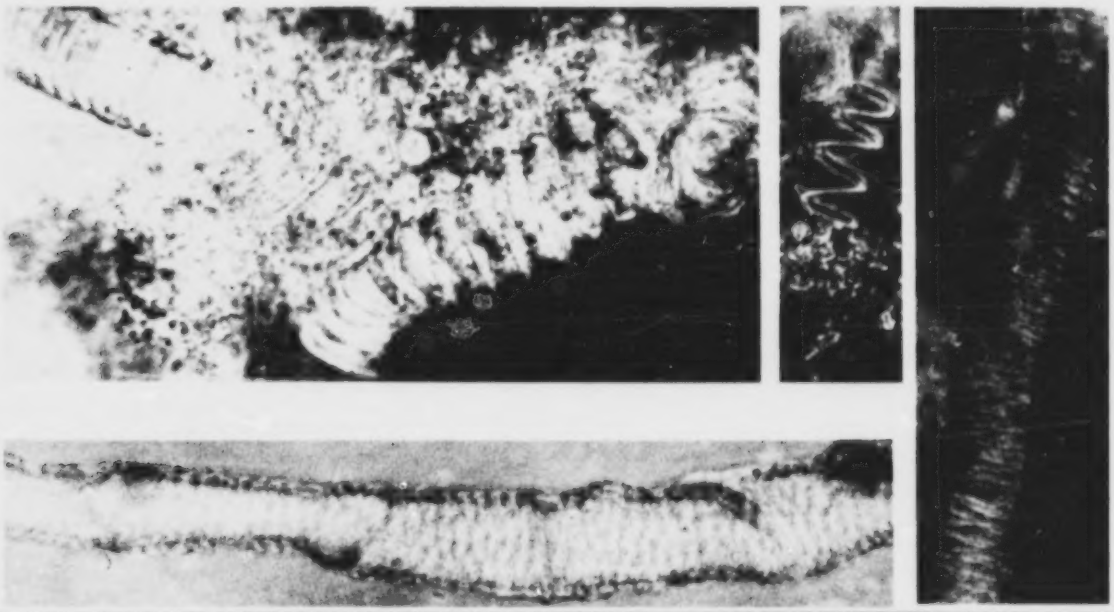
A.

B.

A. Xylem - helical secondary wall thickening
B. As the food substance passes through the digestive and sewage systems to the marine environment, the outer primary wall distintegrates leaving the helical (or spiral) thickenings. These spirals are the identifiers in the RSIT method.

- Sewage material enters the marine environment as discharged raw sewage, as a result of sewage-related accidents, from illegal dumping, or by the purposeful marine disposal of sewage sludge (Figure 5). The remains of the secondary cell walls of terrestrial vascular plants thus enter the marine environment (O'Connor, Stanford, Eds., 1979).

Figure 5: SLUDGE SAMPLE VASCULAR TISSUES



- The helical and spiral patterns of the lignified secondary cell walls of xylem tissues do not naturally exist in the marine environment. Although a few vascular plants such as eel grass and turtle grass exist in the marine environment, they do not exhibit the same structural features as terrestrial plants. "The marine angiosperms all resemble Elodea in lacking true vessels and in being almost totally devoid of any vascular elements with even annular thickenings" (Sculthorpe, 1967, p. 144). The other plants in the marine environment are either microscopic phytoplankton or macroscopic algae, neither of which are vascular plants.

The five foregoing premises are the basis for the following conclusion: that microscopically identified helical or spiral remains of the secondary cell walls of xylem tissue in marine samples indicate the presence of sewage/sludge-related material in the marine environment.

STUDY OBJECTIVES

The NOAA-funded RSIT project was designed to test this conclusion. The objectives of the project were to demonstrate that:

- The indicator is not present in uncontaminated ocean areas.
- Samples from contaminated areas contain the indicator.
- The quantity of indicator present reflects the level of contamination.
- The analysis technique is rapid, cost-effective, and uncomplicated.
- The procedure is effective for fresh, refrigerated, frozen, and formaldehyde-preserved (at least through six years) samples.
- The potential exists for further refinements and applications to the technique.

MATERIALS AND METHODS

In order to validate the technique, 96 water column and sediment samples were collected by the Nassau County Department of Health, Office of Marine Ecology, or provided by the National Marine Fisheries Service of NOAA.

Sample Categories

Samples from a variety of sources were obtained to reflect specific objectives of the project. These categories were:

- uncontaminated and contaminated estuarine and nearshore ocean
- uncontaminated and contaminated ocean
- sewage sludge dump site-New York Bight Apex
- raw and processed sewage from sewage treatment plants

Sample Locations (Figure 6)

- New York Bight Apex - where the dump site is 12 miles off the south shore of Nassau County
- Hempstead Harbor - a north shore harbor of Nassau County and an arm of the Long Island Sound receiving effluent from three sewage treatment plants
- Mid-Atlantic estuaries - samples received "blind", site and category unknown

Figure 6

Sample Collection and Processing

Collection methods and handling varied with sample type. Water samples were either surface dips or taken from one meter above the bottom with a Van Dorn Sampler. All sediment samples were collected with a Smith McIntyre grab. Water samples were either observed fresh, refrigerated, or after preservation with formaldehyde. Sediment samples were either refrigerated, frozen, or preserved with formaldehyde.

Companion Tests

Companion tests for some samples included arsenic, cadmium, chromium, copper, lead, nickel, selenium, zinc, total coliform and fecal coliform (Standard Methods for Examination of Water and Wastewater, 1980).

Plant Analyses

Many of the edible plants typically consumed were sectioned in various planes for photomicrography of the xylem tissue (Figures 2, 3). Marsh plants were studied microscopically to determine if their presence would affect the integrity of the RSIT in nearshore areas. These estuarine plants were tested for lignin content with a phloroglucin and hydrochloric acid staining technique.

Water and Sediment Sample Procedures

For gross screening, a drop of either centrifuged water sample or diluted sediment is observed microscopically. The quantitative procedure is illustrated in Figure 7. Begin at Step 1 for sediment samples and at Step 8a for water column samples.

Figure 7: QUANTITATIVE RSIT PROCEDURE

BEGIN HERE FOR SEDIMENT SAMPLES:

1. Fill 10 ml graduated cylinder with 8 ml of room temperature water (tap).
2. Draw 2 ml of sediment into plastic syringe.
3. Add sediment to cylinder bringing total volume in cylinder to 10 ml.

4. Stopper cylinder and shake by inverting from top to bottom until well mixed (approximately one minute). Remove stopper.

5. Let stand exactly 10 seconds.
6. Immediately draw off supernatant with pipette and reserve.
7. Add water to residue to bring total to 10 ml and repeat steps 4 to 6. Do this twice reserving supernatant.

8. Discard residue and filter reserved supernatant through a 20 micron nytex filter, discarding filtrate.

BEGIN HERE FOR WATER COLUMN SAMPLES:

8a. Filter 500 ml of water through a 20 micron nytex filter, discarding filtrate. (Remainder of procedure is identical for sediment and water column samples).

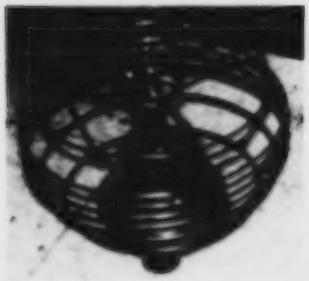
9. Thoroughly flush residue remaining on filter with squirt bottle containing tap water.

10. Wash residue remaining on filter into a beaker using less than 15 ml of water.

11. Pour slurry from beaker into 15 ml centrifuge tube.

Figure 7 continued

12. Centrifuge for one minute forming pellet.



13. Measure and record volume of pellet.

14. Draw off supernatant until double the volume of the pellet is left. For example, if the pellet measures .4 ml, draw off all but .4 ml of supernatant, leaving a total of .8 ml, pellet and supernatant.

15. Stir residue and remaining supernatant with pipette.

16. Using the pipette, place one drop of solution on microscope slide.
17. Add two drops of water.

18. Using probe, mix drops of solution and water to evenly space out residue.
19. Cover with coverslip.

Quantitative Microscopic Technique

- Scan each prepared slide - Under 200x magnification scan for presence of xylem tracheary elements (see Figures 1-5). Phase contrast illumination is preferable. A skilled viewer can benefit from the judicious use of polarized light.

- Count and record the number of xylem tracheary elements on each slide. A cluster of xylem is counted as one. Do not count if identification is uncertain. A number of planktonic plant and animal structures may appear similar to the untrained eye (e.g. various diatom frustules, the myoneme of protozoan stalked ciliates, etc.).

- Follow steps above for three slides from each sample - Average the number of xylem tracheary elements recorded for the three slides. To determine the number per 10 ml use the following calculations:

a. Divide the volume of the pellet obtained in Step 13 by the volume of one drop of solution in Step 16
b. Take the number obtained above, then multiply by the average count of the three slides
c. Double this number to account for the dilution in Step 14
d. The result is the number per 2 ml. To extrapolate to 10 ml multiply by 5

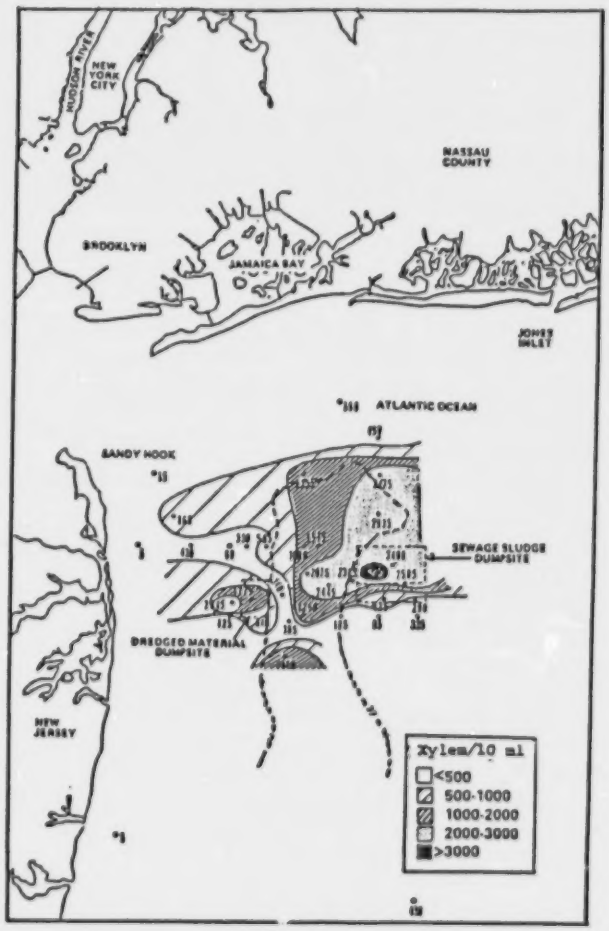
RESULTS AND DISCUSSION

Results from the sediment and water column samples from the ocean and estuarine sites conform to what is known of the study areas. A 1980 study (Reid, et al, 1982) showed sediments with elevated fecal coliform bacteria and Clostridium perfringens spores and concentrations of organic carbon, metals, PCBs, PAHs, and coprostanol all at, or near, maximum values in the area of sewage sludge accumulation just west of the sludge dump site. During the RSIT study, samples near the dump site (except directly south) showed the highest presence of xylem. The sediment samples ranged from 4,875 xylem observations per 10 ml in a dump site sample to none in all the uncontaminated estuarine and nearshore samples.

Ocean Sediment Results

Figure 8

Figure 8 illustrates the xylem concentrations in the 44 New York Bight sediment sample locations. The greatest concentrations are to the north and northwest of the dump site (no samples were collected east of the dump site in this series.) The stations at the southern boundary of the dump site have low xylem concentrations. The xylem concentrations define the pattern of the sludge path to the north and west of the dump site in a crescent shape. A second smaller area of elevated counts coincides with the "cellar dirt" (dredged material) dump site southeast of the sludge dump site. Contamination lessens radially.



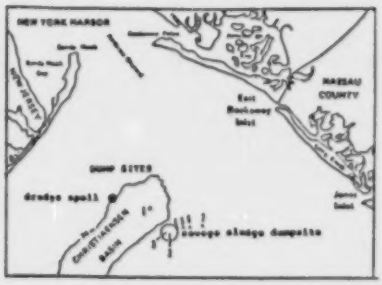
Companion test results are available for seven of these samples (Table 1, Figure 9). The highest xylem counts correspond to the highest fecal and total coliforms, as well as the highest vibrios, cadmium, chromium, copper, lead and

zinc levels. These heavy metals are in essentially the identical rank order as the xylem counts, illustrating that the RSIT is at least as effective as heavy metals in identifying sewage/sludge, with the advantage of immediate results.

Table 1: COMPARISON OF XYLEM COUNTS AND COMPANION TESTS RESULTS FROM SEDIMENT SAMPLES

Station No.	Xylems/ 10 ml	Total Coliform	Fecal Coliform	Vibrio (Bottom Water)	Chemical Analyses (mg/kg dry)						
					As	Cd	Cr	Cu	Pb	Hi	Zn
3	4875	240,000.0	240,000.0	427	12.8	8.5	208.0	338.0	396.0	20.6	511.0
5	2940	240,000.0	110,000.0	144	5.7	3.0	57.3	80.5	92.5	8.2	151.0
4	1620	4,300.0	980.0	113	5.2	2.0	40.6	71.2	89.0	7.9	129.0
1	795	46,000.0	3.0	40	10.9	2.0	29.7	33.3	51.5	49.6	99.0
7	585	46,000.0	290.0	76	17.7	3.0	21.4	12.6	32.0	3.8	102.0
6	540	46,000.0	240.0	124	3.0	3.0	11.6	11.0	27.5	3.5	30.1
2	170	9,300.0	9.1	-	8.8	2.0	12.4	10.0	22.6	6.0	35.8

Figure 9: SEDIMENT SAMPLE SITES

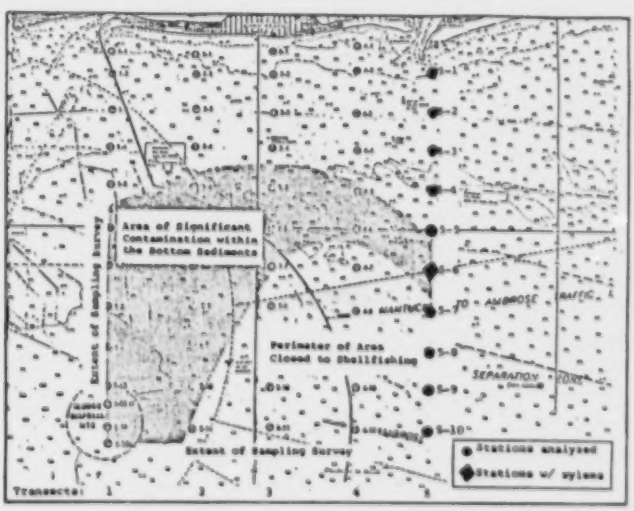


Ocean Water Column Results

During the mid-1970's five transects for water column and sediment sampling were established south of western Nassau County between the dump site and the barrier beach. Three sample sets collected in 1975 and 1978 were examined for presence of xylem. The first set, a transect of 10 bottom stations collected east of the sewage sludge dump site in May, 1975, showed xylem to be present in the one bottom water column sample (5-6) which was judged to be contaminated by traditional tests.

Figure 10: MAY 1975 SAMPLE SITES

Figure 10 illustrates the 5 transects. The 10 samples from transect 5 were the only samples analyzed from this series. Station 5-6 of this easternmost transect is the only station in the transect included in the zone of contamination. This transect, the farthest from the dump site, was originally intended as a control.



In each of the other two series of eight to nine water column samples, the station with the highest xylem count was also the most impacted adjacent sediment station, as shown by companion test results such as heavy metals.

Another valuable result from this samples series is that xylem did persist in formaldehyde for six years.

Raw and Processed Sewage

Xylem elements were observed in all samples from various stages ranging from raw sewage through processed sludge awaiting dumping. The RSIT appears to be effective even after the sludge is mixed with ambient bottom material. No obvious structural changes to the xylem were observed, however, the vascular tissue became progressively freer of surrounding soft tissue and, therefore, easier to distinguish.

Estuaries

Marsh plants were found to be high in lignin, but without observable xylem secondary wall structure, thus eliminating them as a factor in RSIT analysis. The eight estuarine sediment samples were negative for xylem, and of the eight water column samples, only one had any xylem. A second set of estuarine water column samples in the vicinity of a sewage treatment plant showed three samples (of 15) to have low levels of xylem. These three stations and three others had low levels of coliforms. Further sampling is needed in estuaries to verify the value of the RSIT, especially to determine background levels of xylem from terrestrial sources in areas rich with leaf litter.

Future Applications

The advantages of a rapid, low-cost technique for determining sewage/sludge contamination should not be limited to only indicating broad, qualitative levels of contamination. Its potential to quantitatively distinguish between levels of contamination and to distinguish between sources of contamination should be further investigated. By analyzing additional samples from specifically chosen sites with known levels of contamination as shown by traditional tests, a quantitative scale of contamination could be developed which could illustrate the path of the sludge contamination. The technique should prove useful in:

- delineating the fresh impact zone at newly established virgin dump sites such as the 106-mile site south of the New York Bight
- tracking the movement of existing sludge deposits
- tracking the sludge plume released during disposal operations
- monitoring changes or recovery at sites to be abandoned such as the 12-mile dump site in the New York Bight.
- tracking the distribution and abundance of sludge reaching the seafloor where the sludge "accumulation" area is offset from the sludge "dump site" area.
- altering sampling strategy by addition or deletion of stations during sampling cruises based on immediate results of on-board sample screening

- refining sampling protocol by concentrating costlier and more time-consuming sophisticated tests at the positive sites
- allowing more cost-effective use or boat time

CONCLUSIONS

The microscopic presence of xylem tracheary elements can now be used as a valid indicator of sewage/sludge contamination in the marine environment, based on the premise that the fiber content of terrestrial vascular plants used as food is not digested by the human body, that the microscopically identifiable components of this undigested roughage are not normally found in the marine environment and are therefore an indicator of sewage/sludge contamination.

This new technique identifies areas of the marine environment which are impacted by sewage/sludge contamination as well as determines broad categories of sewage contamination.

Because the technique can be used on-site, is rapid and cost-effective, and avoids the delays inherent in traditional testing methods, the RSIT is an important addition to the procedures available for determination of sewage/sludge presence in the marine environment. Its use in estuaries, however, and with water column samples needs further study.

At this point in the refinement of the technique, the greatest benefits of the RSIT are the ease of identification of the indicator and its accuracy in defining the pattern and path of sewage/sludge sediment contamination. This is useful both in designating sampling stations during routine monitoring and in emergency public health situations such as sewage/sludge spills, where the delays in receiving the results of traditional tests are unacceptable for cleanup or beach closure determinations. These are vital inputs to the effective management of marine resources.

A significant factor in the use of the RSIT is the ease with which it can be taught. Training materials should be developed to fully disseminate the RSIT procedures worldwide. With minimal training and access to common laboratory materials, pollution control personnel can conduct their own preliminary analyses and thereby effect management decisions by adding to available information. In times of sewage-related emergency or in vital decision-making processes such as sludge dump site location, effective, valid and timely information is vital and can be augmented by the use of the RSIT.

ACKNOWLEDGEMENTS

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THE ENVIRONMENTAL IMPACT OF TWO DEEP SEA EFFLUENT DISPOSAL SYSTEMS IN NATAL, SOUTH AFRICA

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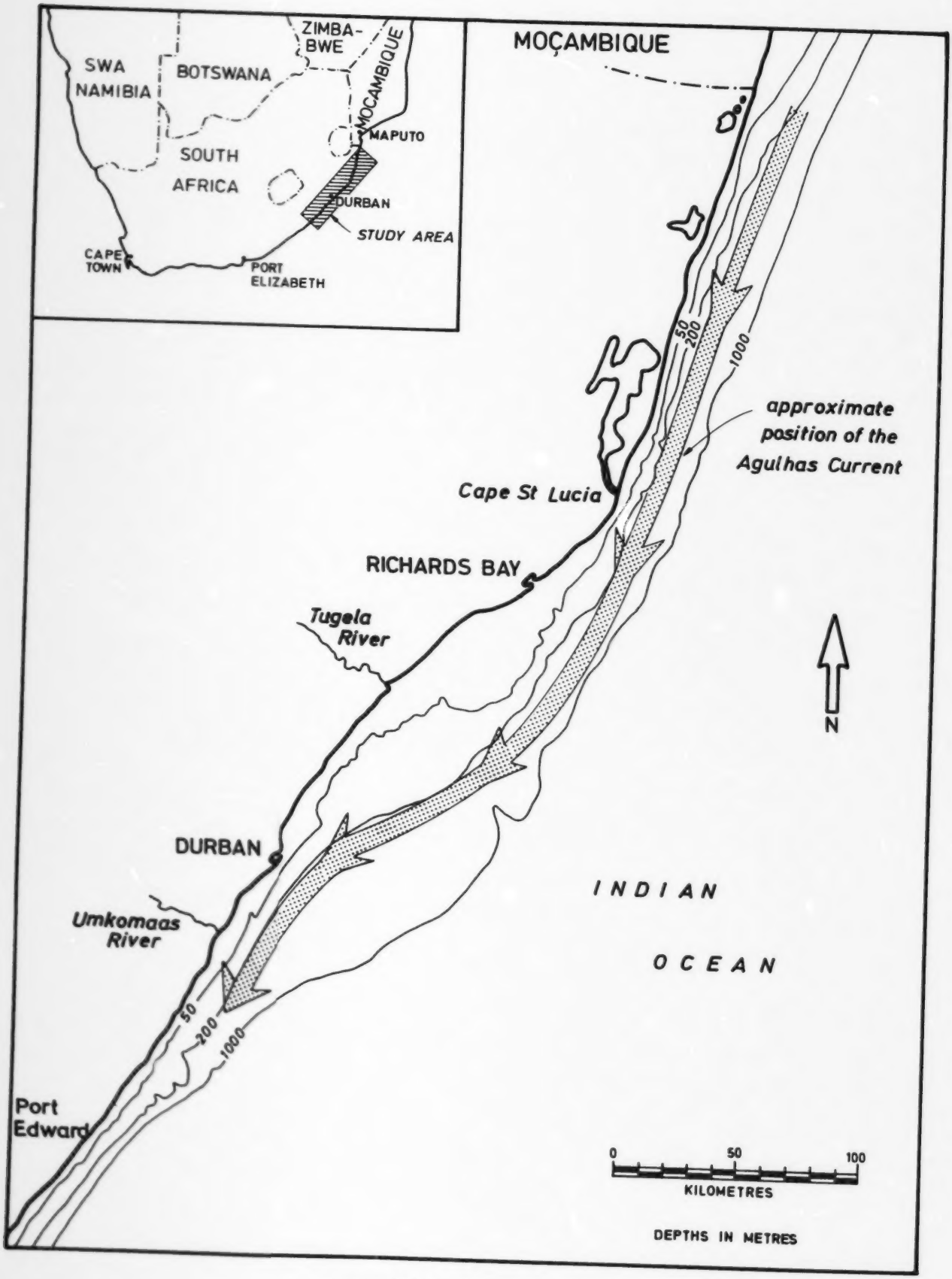
ABSTRACT

Two deep sea effluent disposal systems at Durban and Richards Bay on the east coast of South Africa are described. Results of monitoring surveys to determine their environmental impact are summarised. The surveys, which included faunal, chemical and bacteriological parameters have so far not revealed any major impact. The success of the Durban disposal system in minimising impact can be attributed largely to rapid dilution and dispersal of effluent by strong longshore currents.

BACKGROUND

The coastal zone of Natal on the east coast of South Africa supports a significant tourist industry and the maintenance of healthy oceanic conditions is of prime concern. The two major growth points on the coast, Durban and Richards Bay have constructed regional effluent disposal facilities for discharging selected effluents into the sea.

The greater Durban area supports a population of over one million and contributes nearly 13% of South Africa's gross industrial output. In the late 1960s two deep sea marine outfalls were installed to dispose of sewage and selected industrial effluent. Prior to that the city's effluent reached the sea through a pipe discharging on the


ZIMBA-
BWE
SWA
NAMIBIA
BOTSWANA
MOCAMBIQUE
MAPUTO
SOUTH
AFRICA
DURBAN
STUDY AREA
CAPE
TOWN
PORT
ELIZABETH
MOÇAMBIQUE
50
200
1000
approximate
position of the
Agulhas Current
Cape St Lucia
RICHARDS BAY
Tugela
River
N
DURBAN
INDIAN
OCEAN
Umkomaas
River
50
200
1000
Port
Edward
0
50
100
KILOMETRES
DEPTHS IN METRES

STUDY AREA

surface at the harbour entrance, which operated during outgoing tides, and a series of shore discharges.

The town and harbour at Richards Bay were developed in the 1970s to provide an additional export/import link with the industrialised interior and so take the pressure off Durban; and to provide new facilities for giant bulk carriers. To stimulate industrial development in the region a variety of concessions and incentives are being offered. These include an offshore effluent disposal facility.

THE DURBAN PIPELINES

The two pipes on the sea bed which are known as the Central and Southern Works outfalls, are constructed of bitumen coated steel pipe (1,2 m and 1,37 m diameter) encased in concrete. They are situated about 12 kilometres apart and have 18 and 34 laterally directed ports respectively. Modelling studies showed that effluent dilutions of 300 and 340 times could be expected on the sea surface above the two pipes. The dimensions of the two pipes are given in Table 1.

TABLE 1

DURBAN PIPELINE DIMENSIONS

	Total Length (km)	Diffuser Length (m)	Depth at Discharge (m)	Design Capacity (m^3 day^{-1})
Central Works	3,201	421,8	48 to 53	135 000
Southern Works	4,198	421,8	54 to 64	230 000

Surveys of Durban's bathing beaches and the projected offshore target areas started in 1964, and the "before" picture was established. Following commissioning of the outfalls and the discharge of de-sludged sewage (and subsequently the re-addition of the sludge) intensive monitoring was undertaken to establish the "after" picture. Beach surveys emphasised health aspects while the offshore surveys concentrated on ecological impact. The main findings follow.

Bacteriology

Bacteriological conditions of the bathing beaches improved dramatically with the commissioning of the offshore pipelines. Stringent quality criteria using *E. Coli* 1 counts, parasite units, staphylococci, salmonellas and salinity have shown that potentially contaminative material from the submarine outfalls does not reach the beaches in measurable amounts. Persistent, diffuse, low-level contamination detected near river mouths is attributed to these water courses. Minor transient contamination was detected in the water and sediments adjacent to the offshore diffusers but there have been no permanent, widespread or accumulative effects.

Chemistry

Trace metals showed no sustained upward trend in the offshore disposal area despite there being occasional transitory increases in Hg, Cu, Cd, Pb, Zn and Ni

concentrations in the sediments. Concentrations in the water column remained close to background levels and within acceptable limits.

Occasional low levels of DDT, Dieldrin, PCBs and Lindane were detected in the effluent. However, no significant accumulation of these compounds has occurred in the offshore sediments and biota. Mussels taken from some shore stations yielded increased concentrations of DDT, DDE, TDE and Dieldrin. This is clearly due to land runoff.

Benthic Fauna

Comparisons of benthic faunal communities over the past 20 years did not reveal any major changes attributable to the pipeline discharges. Occasional increases in nematode population density near the diffusers suggest a measure of localised enrichment. However, no sustained decline in species abundance and diversity, nor any suggestion of anaerobic conditions, were detected.

Discussion

The Durban submarine pipelines are proving to be a safe and effective means of effluent disposal. Their efficacy can be attributed very largely to the Agulhas current which promotes rapid dilution and dispersal. Although the discharge areas lie inshore of the main flow of the Agulhas current, clockwise gyres associated with the current produce dynamic dispersal conditions. Bottom

currents near the outfalls tend to flow at about 0,2 m sec^{-1} but may reach 1,0 m sec^{-1}. Surface currents frequently exceed 1,5 m sec^{-1}, and the predominant direction is northwards parallel to the coast.

THE RICHARDS BAY PIPELINES

Industrial effluent and macerated sewage are pumped through two high density polyethylene pipes (A and B) layed in tandem. Pipe "A" (1,0 m diameter) extends further out to sea that "B" and carries a buoyant effluent comprised of paper mill effluent, sewage and fluoridated waste water. This effluent is emitted through 106 vertical ports spaced 6 m apart. Pipe "B" (0,90 m diameter) carries a dense gypsum slurry from a fertilizer factory along with some paper mill effluent and sewage. The effluent is directed upwards, at 30 degrees from the vertical, through 10 jets set 30 m apart. The separation of dense and buoyant effluents allows greater dispersion through natural rising or falling of the effluent after emission. Modelling studies showed that effluent dilutions of 250 and 200 times could be expected above the diffusers of the "A" and "B" lines respectively. Pipeline dimensions are given in Table 2.

TABLE 2

RICHARDS BAY PIPELINE DIMENSIONS

	Total Length (km)	Diffuser Length (m)	Depth at Discharge (m)	Design Capacity (m^3 day^{-1})
"A" Line	4,915	639	27	160 000
"B" Line	3,804	317	23	86 400

The Richards Bay pipelines were commissioned in 1985. Pre-discharge background environmental surveys were made in the period 1974 to 1985. A post-discharge environmental impact assessment is currently being made. A preliminary evaluation follows.

Bacteriology

Classification of waters using *E. Coli* 1 counts, parasite units, staphylococci, salmonellas and salinity revealed a continuing moderate degree of faecal pollution within the bay, while little or no contamination of significance has so far been recorded at beaches to the north and south of the harbour entrance. No significant faecal pollution has yet been recorded in surface water samples taken within 500 metres of the diffusers.

Chemistry

Fluoride and copper concentrations are high in the industrial effluent being discharged at Richards Bay. Stringent efforts are devoted to measuring their possible accumulation offshore. Marginally increased fluoride concentrations have been measured in a few of the post-discharge water samples but there is no evidence of excessive fluoride accumulation in fish taken from the area. Copper concentrations in water, sediments and fish have remained at background levels despite copper concentrations of up to 7 micrograms per litre having been occasionally recorded near the outfall. Mussels transplanted to the discharge zone for two months exposure

showed no significant increase in body burdens of copper. None of the other chemical parameters measured, which included Hg, Cd, Pb, Zn, Fe, Cr, As, Se, DDT, DDE, TDE, Dieldrin, Lindane and PCBs, showed any significant increase above background levels.

Benthic Fauna

Use of the benthic community to assess the environmental impact of the pipelines was severely hampered by the dumping of dredge spoil offshore during harbour construction. Several million tons of primarily silty sediment was dumped close to the study area during the late 1970s. This caused a patchy accumulation of silt on the sandy seabed and a consequent impoverishment of the benthic fauna. The silt has gradually dissipated, and benthic faunal diversity and abundance have steadily increased. Although the pipelines do not appear to have had a detrimental effect on the benthos so far, a wider synoptic survey, planned for 1988, may reveal zones of impact.

Gymsum Accumulation

Although chemically inert, gypsum may act as a smothering agent. Its accumulation on the seabed was, therefore, of prime concern in designing the pipeline. Predictions indicated that, during periods of slack water movement an area of the seabed in excess of 3 hectares would be coated with gypsum. In situ observations made by diving and dredging indicate temporary gypsum

accumulation so far only in the immediate vicinity of the diffusers. However, these accumulations also eventually disperse with changing oceanographic conditions.

Other Parameters

A number of other biological parameters are monitored near the pipelines. These include the neuston and the larger demersal organisms taken by beam trawling. Livers from soles taken in the beam trawls are also examined histologically for possible abnormalities. Results are as yet insufficiently advanced to allow comment.

Discussion

Findings so far indicate no serious impact on this environment. As in the case of Durban, the pipelines appear to be operating as predicted. Close monitoring of the situation will continue as concentrations increase and additional effluents are incorporated. Two areas of particular concern at Richards Bay, when compared with Durban, are the shallowness of the water and the paucity of strong currents to aid effluent dispersal. The shelf break and Agulhas current are considerably further off-shore at Richards Bay. Currents near the diffusers tend to be wind driven and seldom exceed 0,2 m sec^{-1}. Calm conditions with current speeds less than 0,03 m sec^{-1} sometimes last for three days.

ACKNOWLEDGEMENTS

The work summarised here comprises a multi-disciplinary effort over many years. Current contributors at the National Research Institute for Oceanology (Durban Branch) of the Council for Scientific and Industrial Research include D.D. Airey, M.M. Calder, A.D. Connell, D.J. Livingstone, E.A. Lord, T.P. McClurg, N.S. Paynter, P.A. Rathbone, R.R. Sibbald, R.C. Stanton and J.D. Whitehorn. Funds, in addition to those obtained from the Institute, are being derived from the Durban City Corporation and the Department of Environmental Affairs (through the Marine Pollution Section of the South African National Committee for Oceanographic Research).

FISH COMMUNITY STRUCTURE ON A DEEP OCEAN OUTFALL: BARBER'S POINT, OAHU, HAWAII

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New hard bottom substrata afforded by a deep ocean sewer outfall off Oahu, Hawaii attracts large numbers of fish. The adult fish species composition at the outfall diffuser was the same before 1984 during the discharge of low rates (1-3 mgd) of pre-treated sewage as during post 1984 discharge when the effluent was primarily treated (15-19) mgd). However, in 1984 when the rate of pre-treated sewage discharge was increased to 15 mgd, fish community structure changed significantly.

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A COMPARISON OF THE EFFECTS OF DEEP-OCEAN SEWER OUTFALLS ON BENTHIC INFAUNAL COMMUNITIES OF OAHU, HAWAII

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Studies of the benthic communities near three deep-ocean sewer outfalls on the island of Oahu, Hawaii, were carried out to determine the effects of discharged material on the benthic infauna. Two outfalls, Sand Island and Barbers Point, are located on the south shore of Oahu in 72 and 66 m of water, while the third outfall, Mokapu, is located on the east coast of Oahu in 30-33 m of water.

The sediment grain-size distribution was much coarser at the shallower, more exposed Mokapu stations than at the deeper, less exposed Sand Island and Barbers Point areas. Benthic infaunal density and species richness were also much greater at the Mokapu site. Benthic infauna at all study areas were dominated by polychaetes, micromollusks and crustaceans, with crustaceans being far more important at the shallower Mokapu site.

At none of the outfall sites was there any indication of increased organic loads at stations near the Zone of Initial Dilution (ZID) of the diffuser pipe. For all components of the benthos studied, abundance and species richness did not appear related to the presence of the discharge. Cluster analysis of stations confirmed that community composition of the benthic infauna did not differ at stations near the ZID. There was no evidence at the community level of a benthic community stressed by organic material input as described in the model by Pearson and Rosenberg. It is concluded that the effects of the three Oahu deep-ocean sewer outfalls on the benthic infauna are minor.

Introduction

The responses of macrobenthic communities to organic enrichment of the sediments in temperate latitudes are well known. Pearson and Rosenberg (1978) extensively review the effects of organic enrichment on marine benthic systems and list and describe changes in macrofaunal communities which include changes in density, species richness, trophic structure, species composition, recruitment and stress tolerance. Examination of benthic communities in the vicinity of ocean sewage outfalls has suggested that the organic material, together with other pollutants, may result in changes in the benthic assemblages in the vicinity of the outfalls (McNulty, 1961; Smith and Greene, 1976; Dorsey and Synnot, 1980), even those outfalls which are located offshore and in deeper water (Word, 1980; Thompson, 1982). However, little of the work reviewed by Pearson and Rosenberg (1978) was carried out in tropical systems. McNulty (1961) provides data on the effects of sewage pollution on the benthos of tropical Biscayne Bay, Florida. This bay is extremely shallow with average depths ranging only from 1 to 3 m over much of the bay. Wade et. al (1972) have described severe effects of organic pollution on the tropical bottom fauna of Kingston Harbour, Jamaica, which ranges in depth from less than 1m to 18m. However, there is little information available on the effects of ocean sewage outfalls on deeper tropical benthic communities.

Primary treated sewage effluent on the island of Oahu is routinely disposed of via deep-ocean outfall pipes with discharge rates of up to 80 mgd for the Sand Island Wastewater Treatment Plant

which serves the city of Honolulu. During 1986-1987, benthic surveys were carried out for the Sand Island and Barbers Point ocean outfalls on the south shore of Oahu and for the Mokapu ocean outfall on the northeast coast of the island. This paper reports the effects of the sewage discharge on the community structure of the tropical benthic communities in the vicinity of these three deep-ocean outfalls.

Methods

Sampling stations

The specific locations of the sampling stations for each of the three sewer outfall study areas are provided in Figure 1. The spatial location of stations was generally similar among studies in that stations were established within the Zone of Initial Dilution (ZID) of the outfall diffuser pipe, on the boundaries of the ZID, and at reference stations at some distance beyond the ZID.

Seven stations were sampled in the same depth range as the Barbers Point Ocean Outfall Diffuser. Depths at all stations were between 62-72 m. Stations consisted of one location (Z) within the ZID, three stations (B,C,D) on the ZID boundary, one station (A) 2.5 km east of the ZID boundary , one station (E) 0.5 km southwest of the ZID boundary , and one station (F) 2.5 km southwest of the ZID boundary (Figure 1).

For the Sand Island Study, a total of eight stations were established along the isobath equivalent to that of the Sand Island ocean outfall diffuser. Depths for all stations were between 62-84 m. Stations consisted of one location (Z) inside the ZID, two stations at the ZID boundary (NE, NW), one station inshore of the ZID (IS), two stations at 1.2 km from the ZID (IW, IE), and two stations 2 and 5 km from from the ZID (CW, F), respectively (Figure 1).

Six stations were sampled in the same depth range as the Mokapu Ocean Outfall Diffuser. Depths for all stations were between 30-33 m. Stations consisted of one location (Z) within the zone of initial dilution, two stations at the ZID boundary (B1, B2), one station (A) 1

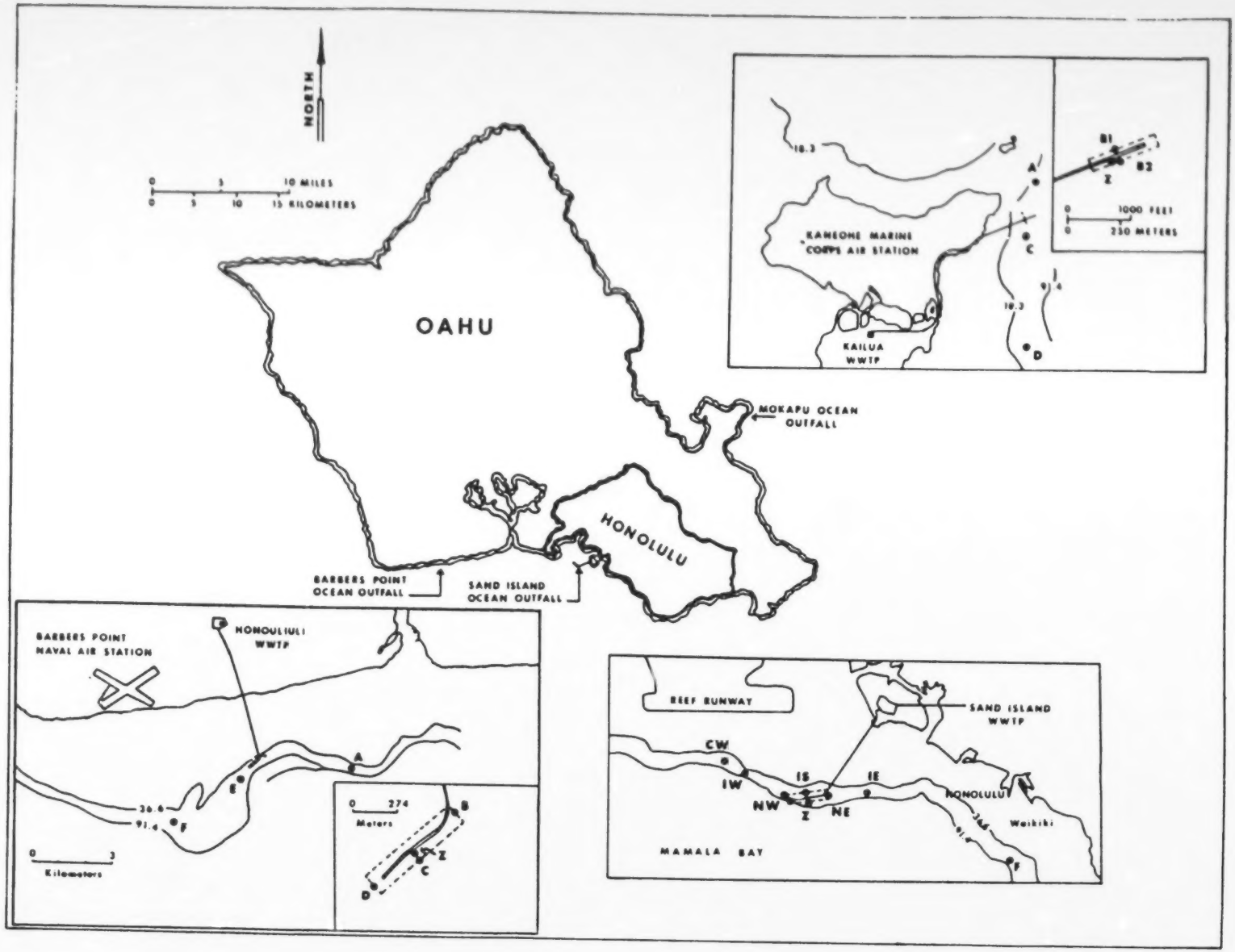

Figure 1. Map of Oahu, Hawaii showing sampling station locations for the Barbers Point, Sand Island and Mokapu deep-ocean sewer outfall benthic studies.

km north of the ZID boundary, one station (C) 1 km south of the ZID boundary (Figure 1).

The exact positioning of each station was determined by a Minolta Mini-Ranger navigation system. Exact station locations (longitude, latitude) are given in several technical reports (Nelson, 1986; Nelson et al. 1987a, 1987b). The precise locations of the ZID and ZID boundary stations for the Sand Island and Barbers Point studies were determined using the submarine Makalii, its tender ship Kila and the City and County of Honolulu (CCH) research vessel Noi I Kai. For the Mokapu study, the location of the diffuser pipe was determined by triangulation on shore targets determined from previous inspection dives. Station locations of the ZID and ZID boundary stations were determined in relation to the diffuser location by divers using compass headings and measured lines. Surface buoys were deployed by the divers and Mini-Ranger positions were determined from the marker buoys.

Sampling methods

A 0.1-m^2 modified Van Veen grab sampler deployed from a stern-mounted A-frame on the RV Noi I Kai was used at all stations of the Sand Island (January 25, 1986) and Barbers Point (September 16, 1986) studies to obtain bottom samples. For both studies, 6 Van Veen grab samples for biological analysis were collected at each station. From each Van Veen grab sample, one 7.6-cm diameter subsample, 5 cm deep was taken. All 6 subsamples were sieved on a 0.5-mm mesh screen while two of the six subsamples were further sieved on a 0.25-mm mesh

screen. Four additional Van Veen grab samples were collected at stations NW, IE and F for the Sand Island study and at stations A and D for the Barbers Point study (Figure 1). These samples were subsampled as above and were sieved on a 0.5-mm mesh screen. Two of the four subsamples were further sieved on a 0.25-mm mesh screen. The remainders of all grabs for biological analysis were sieved on 0.5-mm mesh screen and preserved and archived with CCH.

For the Mokapu study (September 12-13, 1986), SCUBA divers descended to the bottom at each station and collected 6 bottom samples using 275 cc PVC corers (7.6 cm dia. x 6 cm deep) on a predetermined random sampling scheme (Nelson et al., 1987a). Each core was placed in a labeled plastic bag and returned to the surface. Five cores were used for benthic fauna analysis and one core was used for determining sediment total volatile solids and grain size analysis.

For the Sand Island and Barbers Point studies, samples for geological analysis were obtained from the grab samples. Two 7.6 cm diameter subsamples, 5 cm deep, were taken. The top 2 cm of one subsample was used for measurement of total volatile solids, and the top 2 cm of the other subsample was used for grain size analysis. Samples for total volatile solids were placed in screw-cap jars and placed on ice until they could be returned to the laboratory for measurement by loss on ignition. Laboratory analyses of sediment grain size and total volatile solids followed methods suggested by the US Environmental Protection Agency (Tetra Tech, 1986, "Quality Assurance and Quality Control for 301(h) Monitoring Programs: Guidance on Field and Laboratory Methods").

Sample Processing

The samples collected were preserved in individual containers in a 10% formaldehyde solution. Samples were later placed into a large rectangular enamel pan and then elutriated with fresh water following the method of Sanders et al. (1965). The elutriate was poured either through a 0.5-mm screen or through nested 0.5-mm and 0.25-mm screens as called for in the sampling program. Organisms recovered were transferred to individual containers containing a 70% ethanol solution and then stored for later sorting. The remaining sand residue from the elutriation procedure was placed in a plastic bag with a 70% ethanol solution for processing for micromollusks.

Holme and McIntyre (1984) indicate that the efficiency of elutriation procedures runs at approximately 98-100%. For the Barbers Point samples, a portion of the sediment remaining after elutriation was sorted and checked from one replicate of each station. The results (Nelson et al., 1987b) showed that an average of 12% of the total number of individuals (0.5 and 0.25-mm fractions combined) from each station were missed by elutriation, but that these specimens were almost exclusively the sipunculan Aspidosiphon klunzingeri inhabiting old mollusk shells too heavy for removal by elutriation. The elutriation procedure was near 100 % in efficiency for all other species.

The elutriate fraction for each sample was stained with rose bengal for 24 hrs before sorting, washed on a 0.25-mm screen to remove the rose bengal, and then placed in fresh 70% ethanol solution. The sample was then examined under a dissecting microscope, and all

non-micromolluscan organisms were removed. All organisms were identified to the species level whenever possible.

Any sand in the elutriate fraction after the non-micromolluscan organisms had been removed was combined with the other portion of the sand residue for determination of micromollusks. The micromollusks were processed as has been described previously by Kay (Kay, 1975, 1978, 1979a, 1980, 1982; Kay and Kawamoto, 1980, 1983). Sediment samples obtained following the elutriation procedures described above were washed in fresh water and air dried. Aliquots of 5 cm^3 (Sand Island), 10 cm^3 (Barbers Point), or 25 cm^3 (Mokapu) were sorted for micromollusks under a microscope and the picked shells identified to species and counted using Kay (1979b) as the primary taxonomic reference. Only fresh looking shells (shiny surface, with good color, intact) were picked. No distinction between live and dead shell fractions were made and the micromollusk samples therefore represent time averaged samples (Kay, 1975, 1978, 1979a, 1980, 1982; Kay and Kawamoto, 1980, 1983).

Data Analysis

The results of the Kolmogorov-Smirnov test for normality of distribution of the data showed all data sets were normally distributed. All data analysed were tested for assumptions of heterogeneity of variances (F-max test) prior to statistical analysis (Sokal & Rohlf, 1981). Where variances proved heterogeneous, this condition was generally corrected with either the $\log_{10}$ or the $\log_{10}(X+1)$ transformation. Comparisons of stations were made with one-way analysis of variance (ANOVA). Following ANOVA, the a

posteriori Student-Newman-Keuls method was used to determine which differences among stations were significant. If heterogeneity of variances could not be corrected by transformation, the non-parametric Kruskal-Wallis test was used (Sokal and Rohlf, 1981).

An overall comparison of species composition among stations was carried out for the 0.5-mm non-mollusk fraction using cluster analysis (Pielou, 1984). The Bray-Curtis similarity index (Bloom, 1981) (with both untransformed and square root transformed data, Stephenson and Burgess, 1980) and the flexible strategy (beta = -0.25) were used (Boesch, 1977). Computation of the dissimilarity matrix was done using the CLUSTER program developed by the EPA Corvallis Environmental Research Laboratory (Matthews, 1981). To compress the data set to a more manageable size, only those species which contributed at least 0.05% of total abundance were included in the analysis. This resulted in the use of 82 of 162 species from the Barbers Point study, 91 of 126 species from the Sand Island study, and 111 of 229 species from the Mokapu study.

Results

Sediment grain size showed relatively little variability among Barbers Point stations (Figure 2). For the finest sediments in classes 3-8 phi (250 - 4 microns), composition ranged only between 1.7 and 6.0%. Station Z had a somewhat lower percentage of larger sized particles (-1 to 0 phi, 2 to 1 mm) than the other stations, but had a similar percentage of finer sized particles.

Sediment grain size was similar at all Sand Island stations except NE which had relatively coarser sediment and F which had relatively finer sediment than the other stations. Sediments at all stations were composed of primarily sand sized particles (Figure 2).

Sediment grain size showed some variability among Mokapu stations. Stations A, B2, C were all very similar in terms of their sediment grain size distribution (Figure 2). Station B1 had very little fine sediment in classes 3-8 phi (250 - 4 microns), while the larger size classes were relatively similar to stations A, B2, and C. Station Z had a lower percentage of larger sized particles than this group of stations, but had a similar percentage of very fine sized particles.

Percentages of total volatile solids in the sediments were similar at all Sand Island stations except F which had a slightly higher percentage (Table 1). Values for total volatile solids ranged between 4 and 5.6 per cent. At the Mokapu stations, percentages of total volatile solids in the sediments were somewhat lower at the ZID stations (B1, Z, B2) than at the other stations (Table 1). At the Barbers Point stations total volatile solids were similar between the

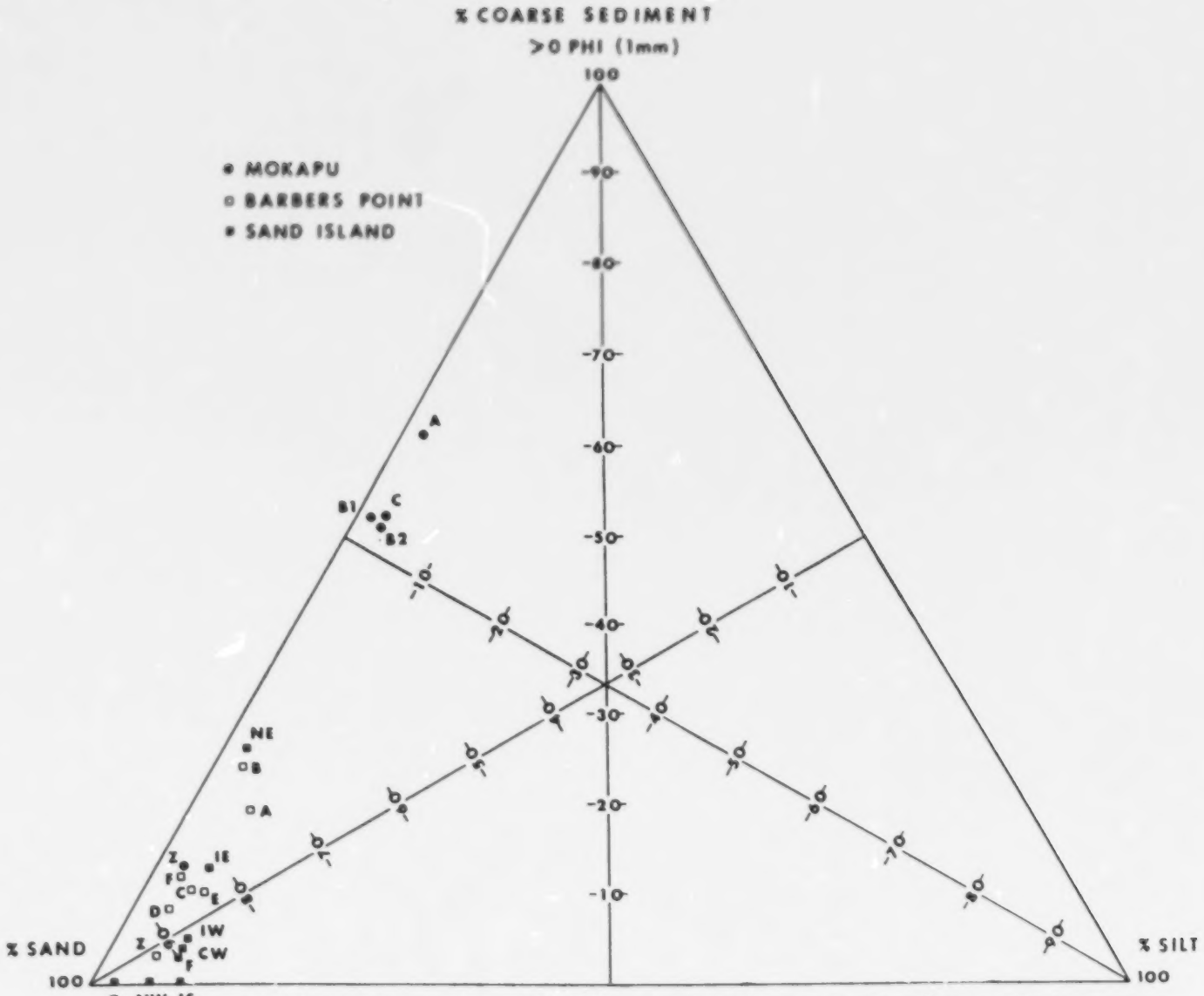

Figure 2. Comparison of sediment composition expressed as percent coarse sediment (> 0 phi), per cent sand (1 - 4 phi) and per cent silt (< 4 phi) among sampling stations for the Barbers Point, Sand Island and Mokapu deep-ocean sewer outfall benthic studies.

Table 1. Comparison of per cent total volatile solids in the sediments between stations located near the Zone of Initial Dilution (ZID) of the outfall diffuser and reference stations for three Hawaiian deep-ocean sewage outfalls.

% Total Volatile Solids	Near-ZID Stations				Reference Stations			
Barbers Point Outfall								
	B	C	Z	D	A	E	F	
	1.8	2.0	2.5	2.4	2.4	1.7	2.1	
Sand Island Outfall								
	IS	NE	Z	NW	F	IE	IW	CW
	4.7	4.6	4.5	4.2	5.4	4.5	5.0	4.2
Mokapu Outfall								
	B1	Z	B2		A	C		
	2.8	2.7	2.9		3.3	3.8		

ZID and ZID-boundary stations (B, Z, C, D) and the other stations (Table 1), ranging only between 1.7 and 2.6%. There was no indication that the stations near the ZID possessed a higher percentage of total volatile solids than other stations at any of the three outfall study areas. Therefore there was no indication of an organic enrichment of sediments near the ZID as compared with more distant locations.

Barbers Point Outfall Study

One hundred and sixty-two non-mollusk and 121 mollusk (primarily gastropod) taxa were collected. Arthropods contributed 41 taxa, with the amphipods being represented by 14 taxa. Other arthropod groups (27 taxa) included isopods, tanaidaceans, cumaceans, decapods, mysids, copepods, acarinids and insect larvae. Polychaetes had the highest number of species of any non-mollusk group with 97, while oligochaetes, nemerteans, sipunculids, echinoderms and chordates constituted the remaining non-arthropod taxa. A complete list of abundances for each species from each replicate for each faunal fraction is given in the technical report describing this study (Nelson et al., 1987b).

For the 0.5-mm non-mollusk fraction, mean abundance ranged between 79.5 (B) and 172.5 (Z) animals per core (Figure 3). Mean abundance for the 0.25-mm non-mollusk fraction ranged from 64 (B) to 192.5 (E) animals per core (Figure 3), with the minimum value being found at the same stations as for the 0.5-mm non-mollusk fraction. Mean abundance of the 0.25-mm non-mollusk fraction was less than that of the 0.5-mm non-mollusk fraction at stations A, B, F and Z and was higher than that of the 0.5-mm non-mollusk fraction at stations C, D

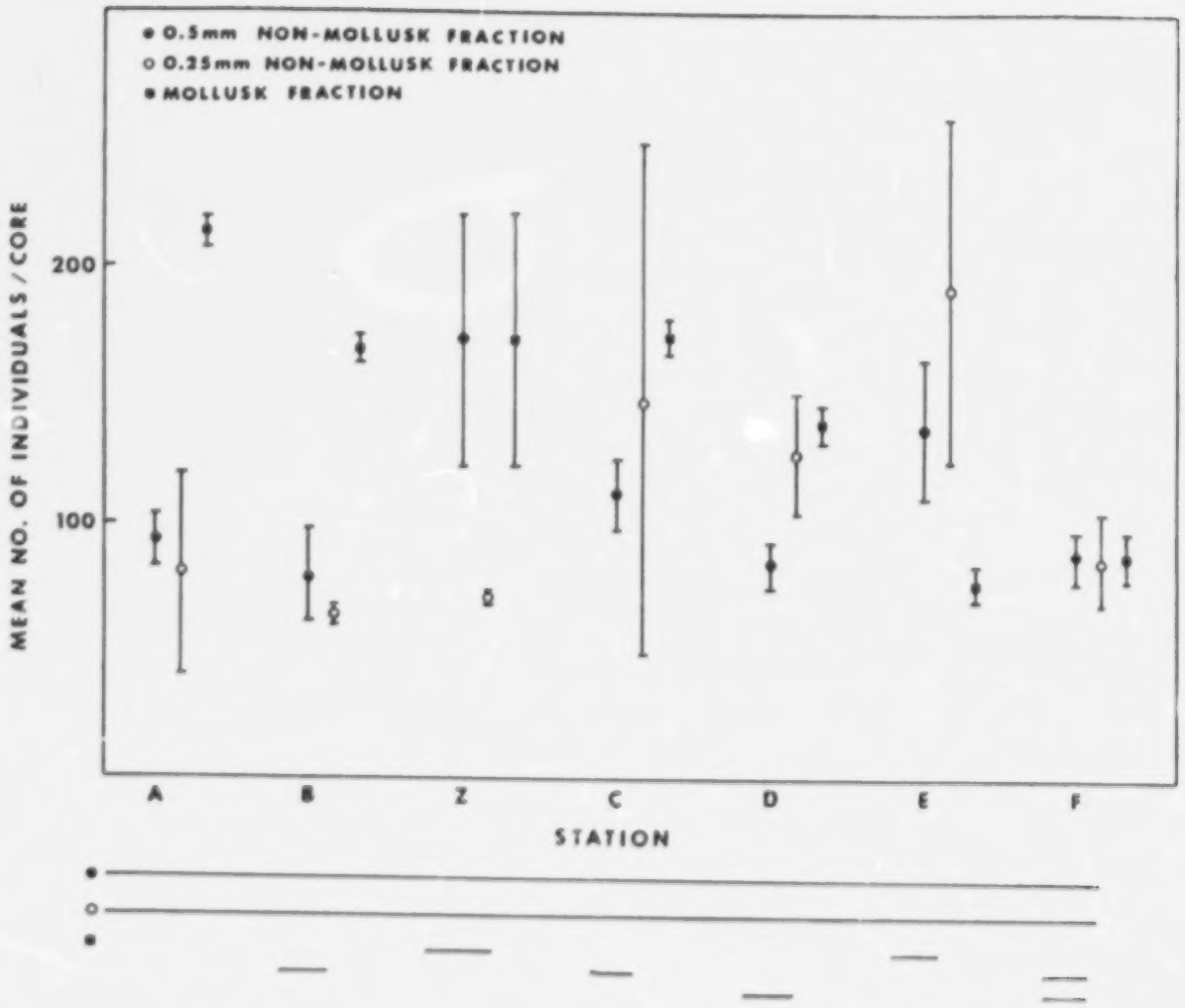

Figure 3. Mean (± 1 std. error) number of individuals per core at each station for the 0.5-mm non-mollusk fraction, 0.25-mm non-mollusk fraction and the mollusk fraction for the Barbers Point ocean outfall benthic study.

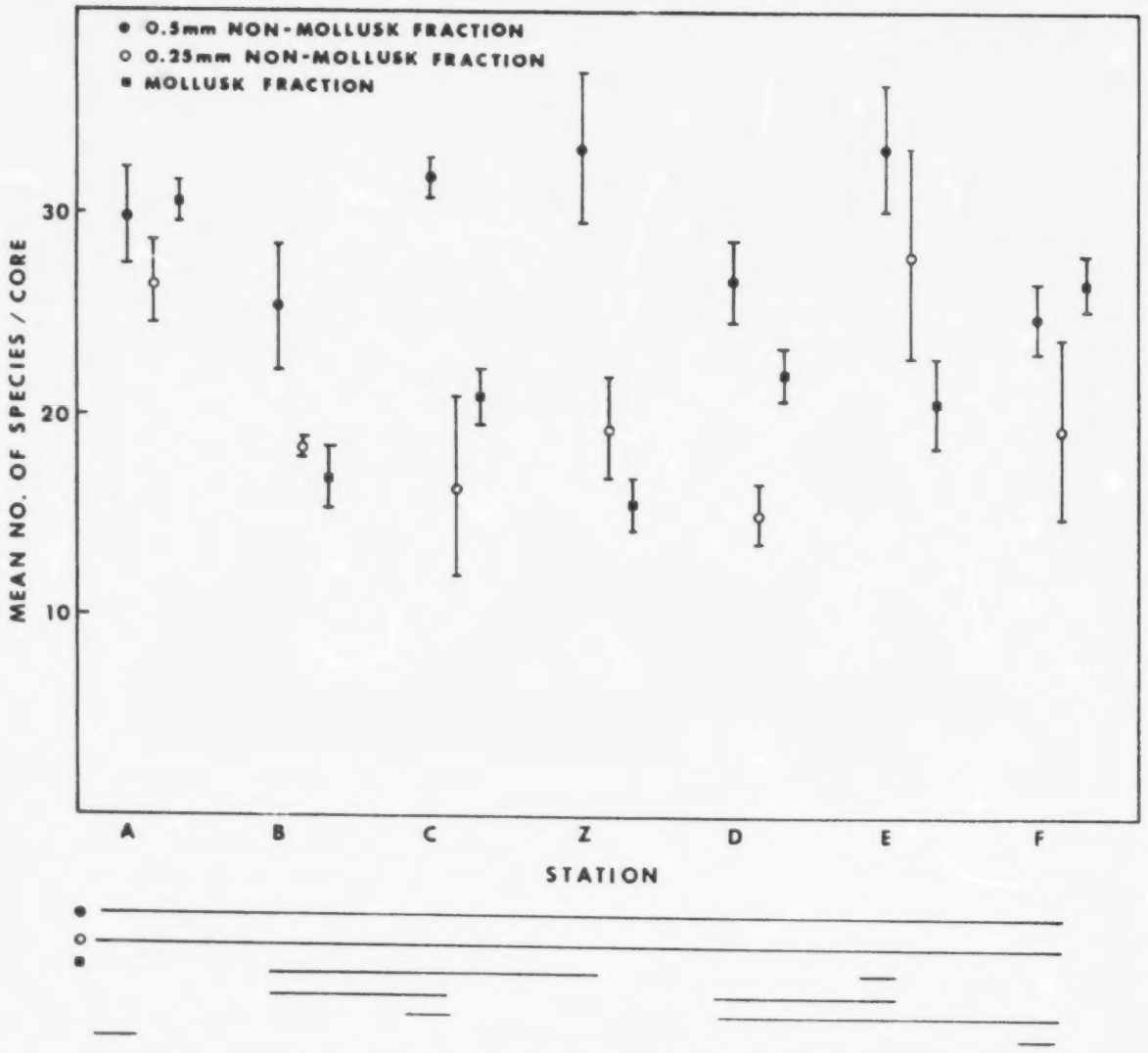

Figure 4. Mean (± 1 std. error) number of species per core at each station for the 0.5-mm non-mollusk fraction, 0.25-mm non-mollusk fraction and the mollusk fraction for the Barbers Point ocean outfall benthic study.

and E. Mean total mollusk abundance ranged between 87 (E) and 213.5 (A) shells per mollusk sample (Figure 3).

Mean number of non-mollusk species ranged between 25.3 (B) and 33.3 (E, Z) per core for the 0.5-mm fraction (Figure 4). Mean number of non-mollusk species ranged between 15.3 (D) and 28 (E) per core for the 0.25-mm fraction (Figure 4). Mean number of species per core was lower for all stations in the 0.25 mm-fraction. Mean number of mollusk species per sample ranged from 15.8 (Z) to 30.5 (A) (Figure 4).

Analyses of differences among stations for numbers of individuals for both the 0.5-mm non-mollusk and the 0.25-mm non-mollusk fractions ($\log_{10}$ transformed) indicate that there were no significant differences among station mean abundances for either size category (Figure 3). Highly significant ($p<0.001$) differences among stations in mean mollusk abundance per station were present. Stations E and Z were not significantly different from one another. Two overlapping groups composed of stations not significantly different from one another were made up of stations D and F and stations F, B and C. Only station A was significantly different from all other stations.

No significant differences among stations in mean number of species for either the 0.5-mm or 0.25-mm non-mollusk fractions were found (Figure 4). For the mollusk fraction, a highly significant ($p<0.001$) difference among stations was indicated by ANOVA. A posteriori comparison of means showed that these differences among stations were largely overlapping. Non-significant differences were indicated for groups consisting of the following stations: stations Z, B, E, C; stations B, E, C, D; stations E, C, D, F; and stations F and

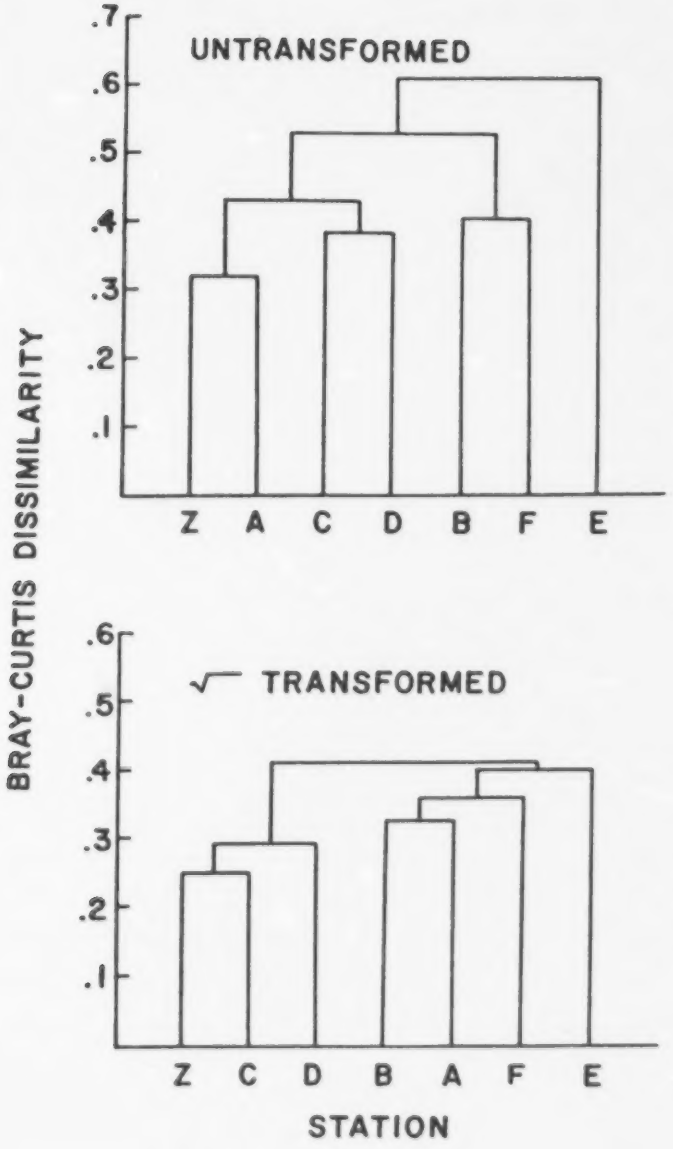

Figure 5. Dendrograms for untransformed and square root transformed data for the 0.5-mm non-mollusk fraction showing dissimilarity among Barbers Point ocean outfall stations.

A (Figure 4). Stations F and A had a higher number of species than stations Z and B. However, since station E did not differ from station Z or B, there was not a strong statistical case for an outfall effect on number of mollusk species.

Comparison of overall species composition and abundance among stations for the 0.5-mm non-mollusk fraction by using the Bray-Curtis index of dissimilarity and cluster analysis confirms the patterns found by ANOVA (Figure 5). The untransformed dendrogram indicates a successive clumping of pairs of stations with no pattern relating order of clumping to station position in relation to the ZID. For example, stations Z and A are grouped together, as are stations B and F. Station E is the most dissimilar to other stations. For the square root transformed data, two groups are joined at a dissimilarity value of only 0.41, indicating all stations are relatively similar in species composition. In this case three ZID and ZID-boundary stations are grouped together (Z, C, D), while one ZID-boundary station (B) is grouped together with the reference stations (A, F, E).

Sand Island Outfall Study

All stations were dominated by polychaete worms in terms of both numbers of individuals and number of species while arthropods were relatively uncommon at all stations (Nelson, 1986). Polychaetes contributed 68 species while oligochaetes and amphipods contributed 13 and 12 species to the total of 126 species in the non-mollusk component. There were 157 mollusk species collected, the great majority of which were gastropods (Nelson, 1986). Complete abundance

and species composition data are available in the technical report on this study (Nelson, 1986).

Mean abundance for the 0.5-mm non-mollusk fraction ranged only between 66.5 (F) and 166 (Z) animals per core, with the exception of station CW (453 /core, Figure 6). The high abundance at CW was due to the occurence of one polychaete, Euchone sp. B. Mean abundance for the 0.25-mm non-mollusk fraction ranged between 21.5 (F) and 111 (CW) animals per core (Figure 6). Mean mollusk abundance ranged between 83 (NW) and 253.3 (IW) per core (Figure 6).

Mean number of species ranged between 16.3 (F) and 29.5 (NE) per core (Figure 7) for the 0.5-mm non-mollusk fraction. Mean number of species ranged between 7 (F) and 17.5 (IS, NE) per core for the 0.25-mm non-mollusk fraction (Figure 7). For the mollusk fraction, mean number of species per core ranged from 16 (NW) to 30.8 (IS).

For mean numbers of individuals for the 0.5 mm non-mollusk fraction, ANOVA indicated highly significant differences ($p<0.001$, log transformed data) among stations. However, a posteriori comparison of means (Figure 6) indicated that this results from the fact that abundance at station CW was greater than all other stations, none of which were significantly different from each other. Abundance of the 0.25-mm non-mollusk fraction showed significant differences among stations ($p<0.05$), but the groupings of non-significantly different stations were broadly overlapping and included both ZID and reference stations (Figure 6). Significant differences ($p<0.05$) among stations were also found for the mollusk fraction. Station IW had significantly more mollusk shells than did the other stations (Figure

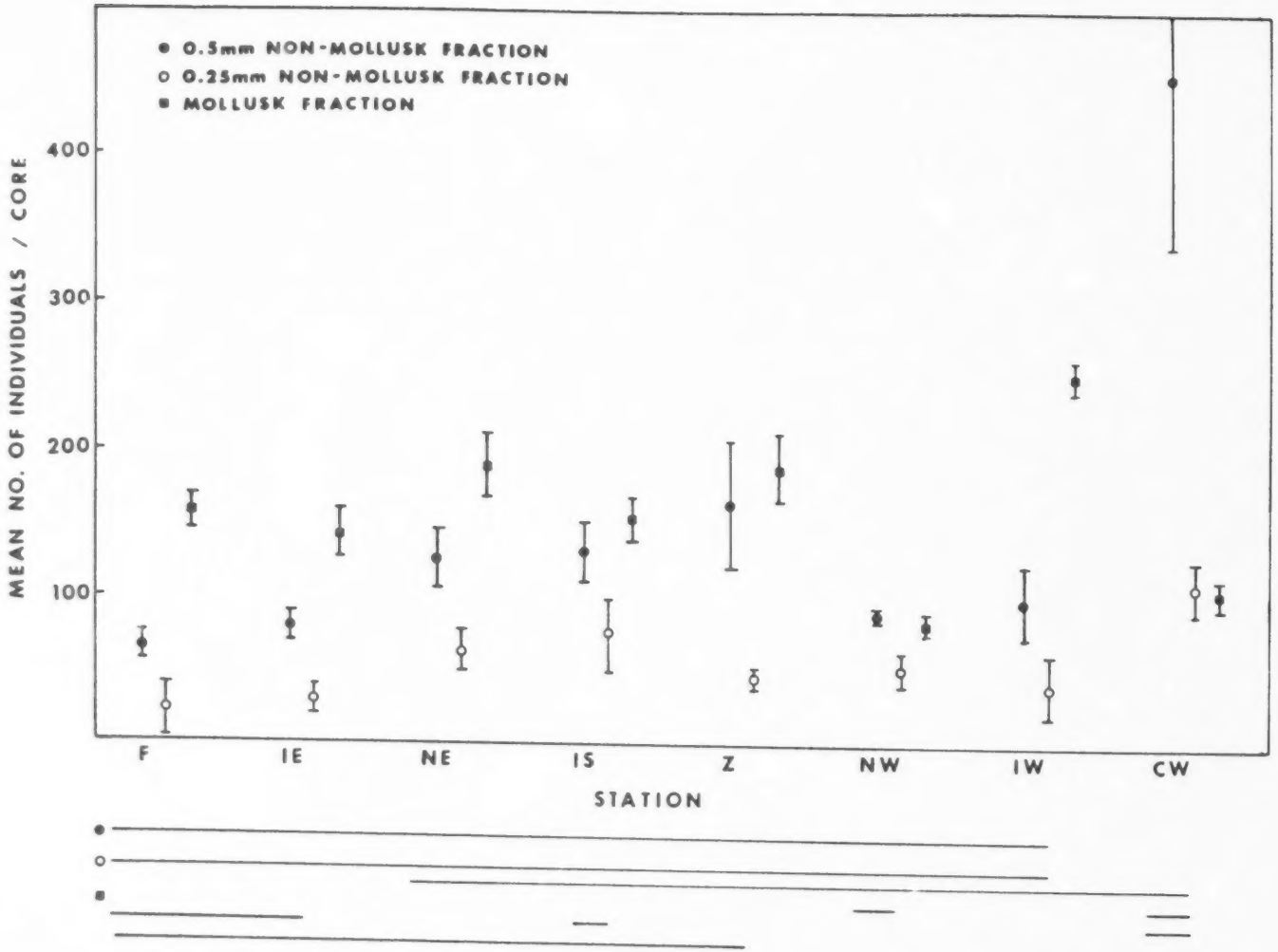

Figure 6. Mean (± 1 std. error) number of individuals per core at each station for the 0.5-mm non-mollusk fraction, 0.25-mm non-mollusk fraction and the mollusk fraction for the Sand Island ocean outfall benthic study.

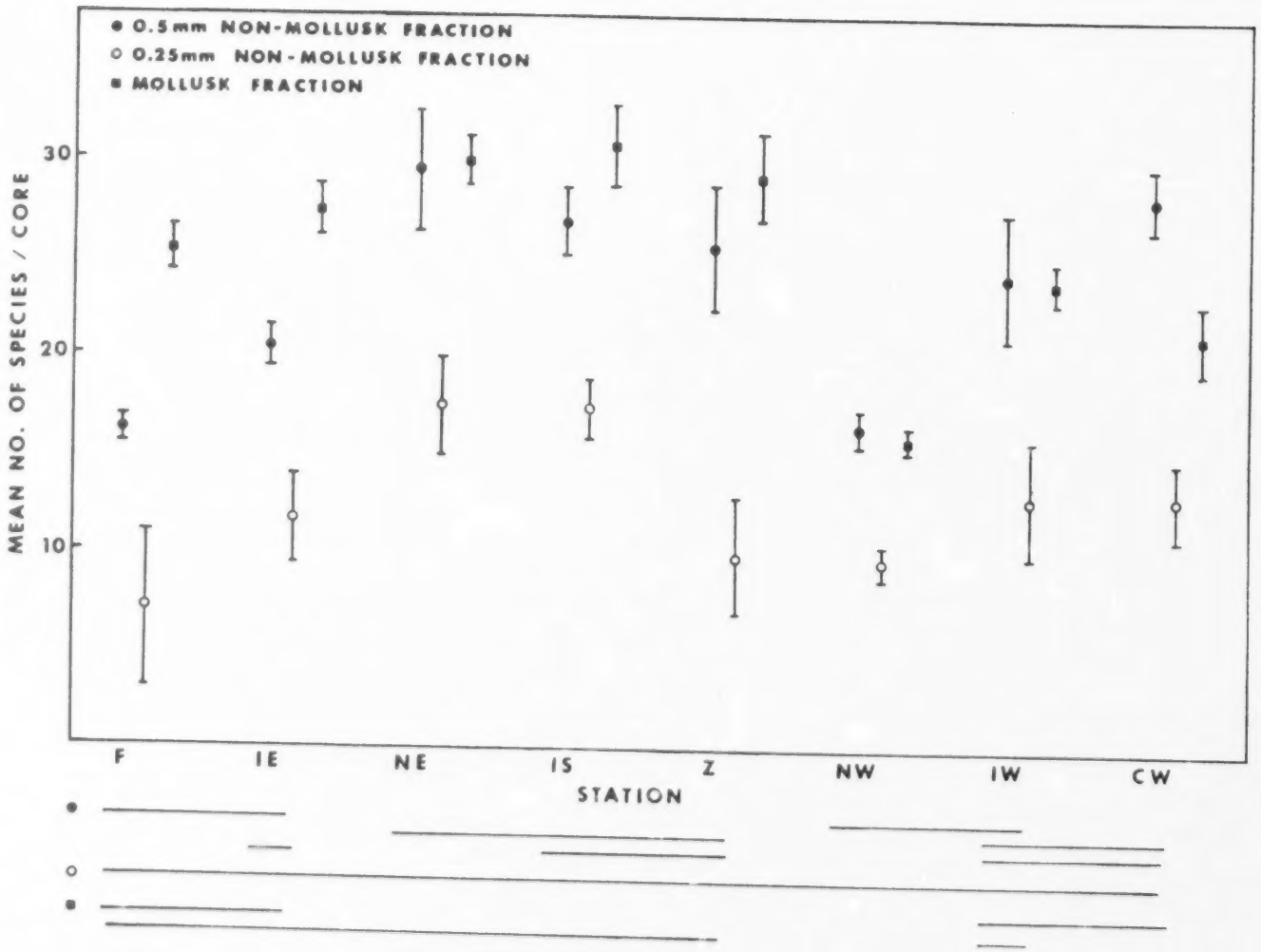

Figure 7. Mean (± 1 std. error) number of species per core at each station for the 0.5-mm non-mollusk fraction, 0.25-mm non-mollusk fraction and the mollusk fraction for the Sand Island ocean outfall benthic study.

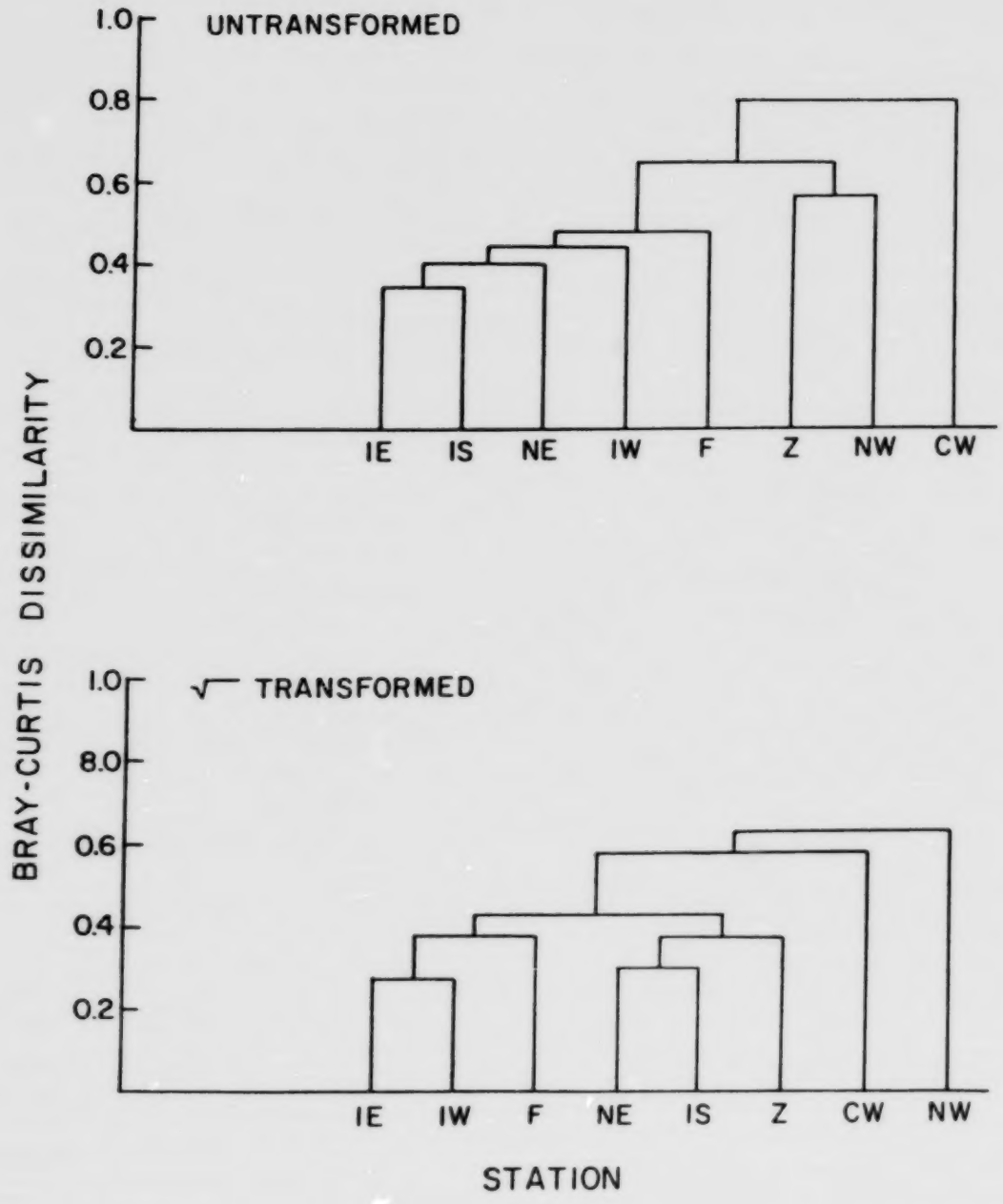

Figure 8. Dendrograms for untransformed and square root transformed data for the 0.5-mm non-mollusk fraction showing dissimilarity among Sand Island ocean outfall stations.

6). The remaining stations formed broadly overlapping groups of stations non-significantly different from one another (Figure 6).

Highly significant differences ($p<0.001$) in mean number of species among stations for the 0.5-mm non-mollusk fraction (Figure 7) were found. Again, a posteriori comparison among means showed broadly overlapping groups with no pattern of significant differences for near-ZID stations as compared with other stations (Figure 7). There was no significant difference among stations for the 0.25-mm non-mollusk fraction. For the mollusk fraction, station NW on the western ZID boundary had significantly fewer ($p<0.05$) species than all other stations. The remaining stations again formed broadly overlapping groups (Figure 7). Stations CW, F, IW, and IE were not significantly different from one another. Stations F, IW, IE, Z, NE and IS were also not significantly different from one another.

Comparison of overall species composition and abundance among stations using the Bray-Curtis index of dissimilarity and cluster analysis confirmed that stations are generally similar to one another. The dendrogram for untransformed data showed station CW to be most dissimilar from the other stations (Figure 8). Where data were square root transformed to compensate for the effects of dominant organisms on the index, stations IE, IW, F, NE, IS, and Z were all grouped as very similar, with stations CW and NW only slightly less similar.

Mokapu Outfall Study

The benthic fauna of the Mokapu study area is both abundant and extremely diverse. Two hundred twenty-nine taxa were collected in the non-mollusk fractions of the samples. Arthropods contributed 105 taxa, with the amphipods and isopods being represented by 42 and 21

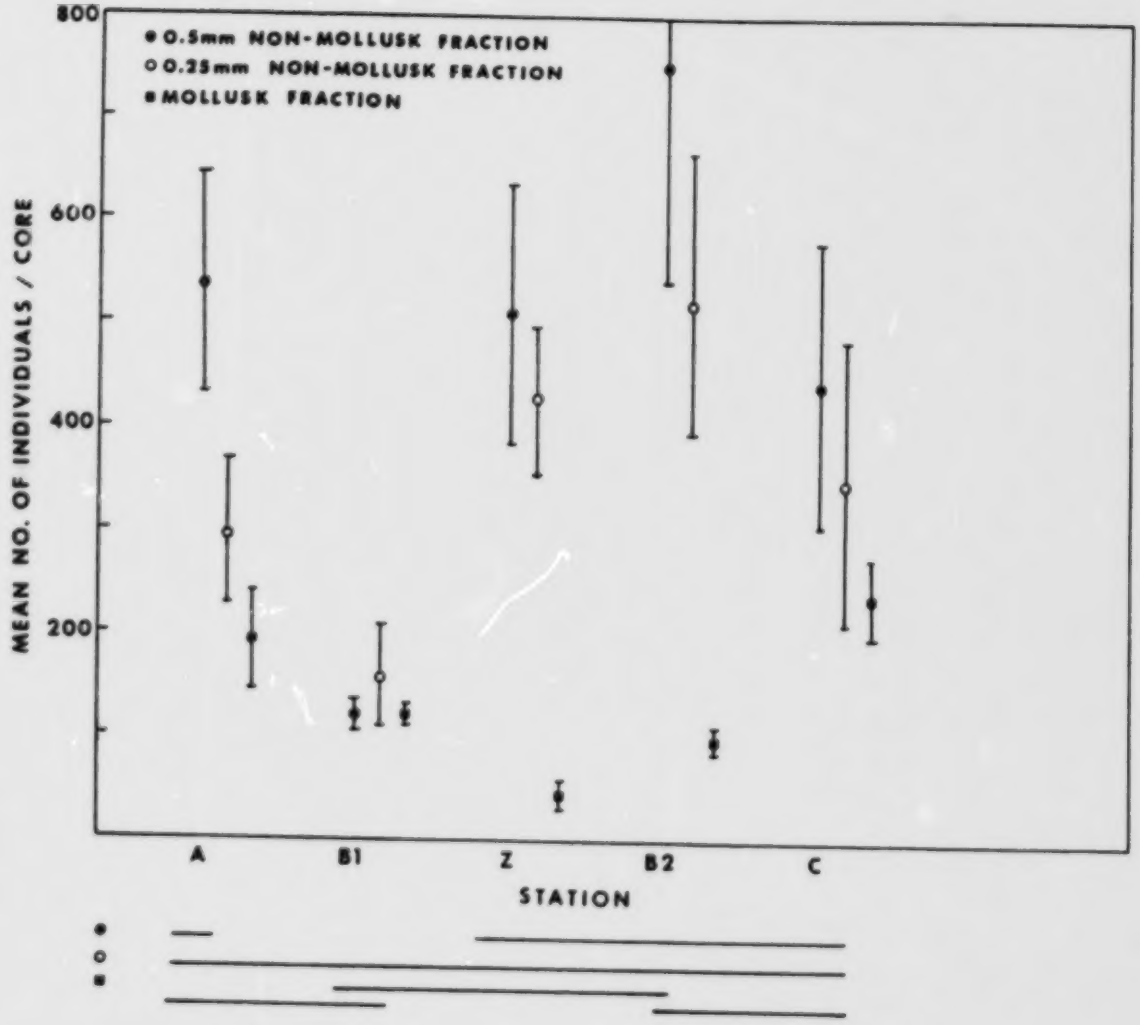

Figure 9. Mean (± 1 std. error) number of individuals per core at each station for the 0.5-mm non-mollusk fraction, 0.25-mm non-mollusk fraction and the mollusk fraction for the Mokapu ocean outfall benthic study.

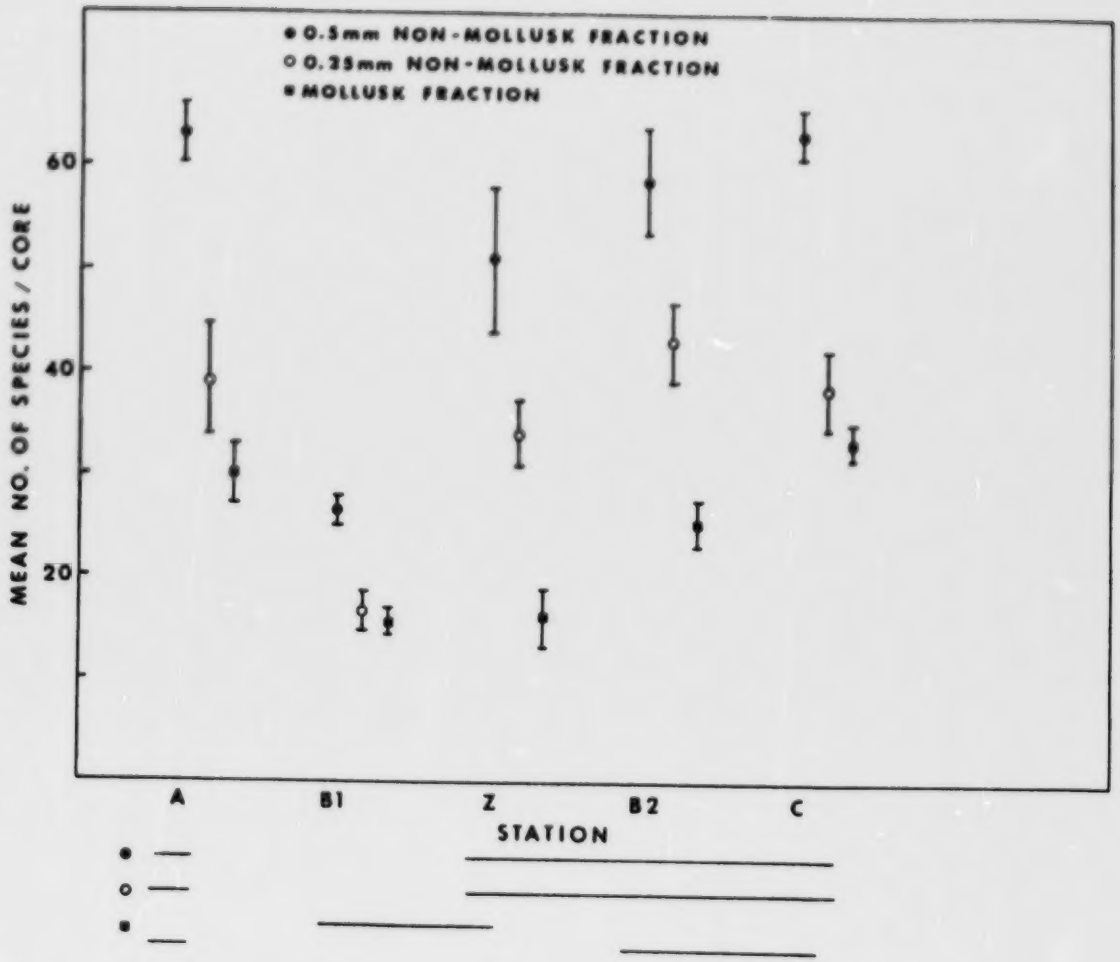

Figure 10. Mean (± 1 std. error) number of species per core at each station for the 0.5-mm non-mollusk fraction, 0.25-mm non-mollusk fraction and the mollusk fraction for the Mokapu ocean outfall benthic study.

taxa, respectively. Polychaetes had the highest number of species of any non-mollusk group with 93, while oligochaetes, nemerteans, sipunculids, echinoderms and chordates constituted the remaining taxa. For the total mollusk fraction, 122 taxa were collected at the Mokapu study area. The large majority of taxa collected were gastropod rather than bivalve mollusks. Complete abundance and species composition data are available in the technical report on this study (Nelson, 1987b).

For the 0.5-mm non-mollusk fraction, mean abundance ranged between 121 (B1) and 756.2 (B2) animals per core (Figure 9). Mean abundance for the 0.25-mm non-mollusk fraction ranged from 158 (B1) to 533 (B2) animals per core (Figure 9), with minimum and maximum values being found at the same stations as for the 0.5-mm non-mollusk fraction. Mean abundance of the 0.25-mm non-mollusk fraction was less than that of the 0.5-mm non-mollusk fraction at stations Z, B2 and C and higher than that of the 0.5-mm non-mollusk fraction at stations A and B1. Mean total mollusk abundance ranged between 46.4 (Z) and 238.8 (C) shells per subsample (Figure 9).

Mean number of non-mollusk species ranged between 26.6 (B1) and 63.2 (A) per core for the 0.5-mm fraction (Figure 10). Mean number of non-mollusk species ranged between 16.8 (B1) and 49.3 (B2) per core for the 0.25-mm fraction (Figure 10). Mean number of species per core is lower for all stations in the 0.25-mm fraction. Mean number of mollusk species per subsample ranged from 15.8 (B1) to 33.8 (C) (Figure 10).

Comparison of means for number of individuals among stations for the 0.5-mm non-mollusk fraction indicated a highly significant difference among stations ($p<0.01$, log transformed). The abundance at

station B1 is significantly lower than for the other stations, which are not significantly different from one another (Figure 9). Analysis of differences for numbers of individuals among stations for the 0.25-mm non-mollusk fraction found no significant differences for this size category. Abundance data for the mollusk fraction showed highly significant ($p<0.001$, Kruskal-Wallis test) differences among stations. Stations Z, B2 and B1 were not significantly different from one another, while an overlapping group of stations B2, B1, A and C were not significantly different from one another. Therefore, only station Z is significantly different from stations A and C (Figure 9).

For number of species in the analysis for the 0.5-mm non-mollusk fraction, a highly significant difference ($p<0.001$) among stations was found, with station B1 possessing a lower value of mean number of species than the other four stations, which were not significantly different from one another (Figure 10). Analysis of differences in mean number of species for the 0.25-mm non-mollusk fraction indicated that station B1 had a highly significant ($p<0.001$), lower mean number of species than the other stations (Figure 10). Stations Z, C, A and B2 did not differ significantly for this parameter. For the mollusk fraction, a highly significant ($p<0.001$) difference among stations was indicated by ANOVA. Stations B1 and Z were found to be similar to each other in mean number of species per core, while both had significantly lower values than stations B2, A and C, which were not significantly different from one another (Figure 10).

Comparison of overall species composition and abundance among stations using the Bray-Curtis index of dissimilarity and cluster

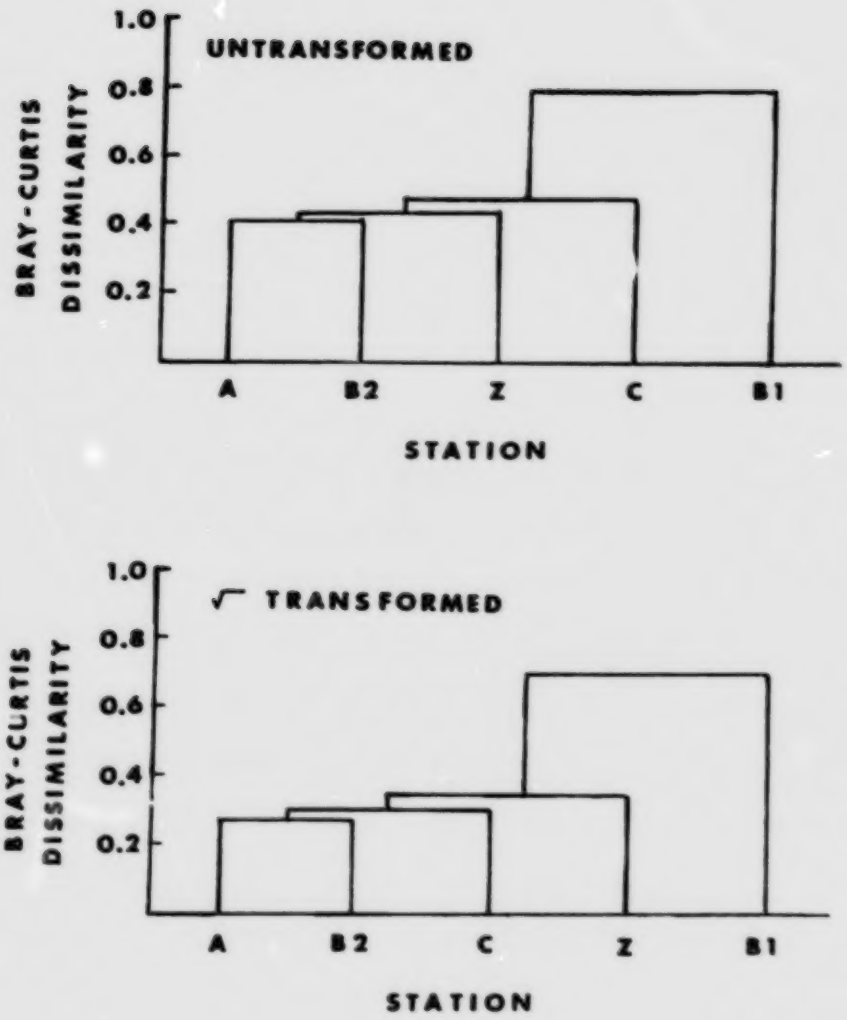

Figure 11. Dendrograms for untransformed and square root transformed data for the 0.5-mm non-mollusk fraction showing dissimilarity among Mokapu ocean outfall stations.

analysis confirmed the patterns found by ANOVA. Cluster dendrograms for both untransformed and square root transformed data both indicated that stations A, B2, Z and C were very similar to one another, while station B1 was more dissimilar (Figure 11). The two dendrograms differed only in the order in which stations Z and C were clustered.

DISCUSSION

The faunal composition and abundances of individuals recorded in the Barbers Point study are much more similar to those of the Sand Island area than the Mokapu area. For example, polychaetes dominated both abundance and number of species at both Barbers Point and Sand Island, while arthropods were much more abundant at Mokapu. The number of polychaete species recorded at Barbers Point (97) was greater than that recorded from either the Sand Island area (68) or the Mokapu area (93). Number of oligochaete species was more similar in two south shore studies (11, 13) as compared with the north shore Mokapu study (6). Number of amphipod species was also more similar for Barbers Point and Sand Island (14, 12) than for Mokapu (42). The range of abundance in mean number of individuals per station was 79 to 172 for Barbers Point stations and 66 to 453 for Sand Island stations, whereas mean abundance tended to be higher and ranged from 121 to 756 at the Mokapu stations.

The differences and similarities among study areas would be expected since the Barbers Point study area and the Sand Island study area are both on the more protected south coast of Oahu and are located in a similar depth range. Bottom topography and sediments at the Barbers Point study area are both more similar to the Sand Island study area than they are to the Mokapu study area. Because of the shallower depth and greater degree of exposure to storm currents and waves, sediment grain size distributions at the Mokapu stations are much coarser than both the Barbers Point and the Sand Island areas.

The shallower depth of the Mokapu stations probably allows an increased level of algal production, which supports the higher density and species richness of crustaceans, particularly amphipods, which were found at the Mokapu stations.

The typical patterns of alteration in the composition of marine benthic communities as the result of excessive inputs of organic matter such as sewage have been summarized by Gray (1981) and analysed in detail by Pearson and Rosenberg (1978). Typically the effect of the most extreme organic input is the total lack of oxygen and the presence of hydrogen sulphide in the sediments together with an absence of benthic macrofauna. With increased distance from the input source, abundance of organisms typically rises sharply to the maximum values found on the gradient. The abundance maximum is composed of one or two extremely abundant, opportunistic species and is termed by Pearson and Rosenberg (1978) the "peak of opportunists". With still greater distance from the input source, a "transition zone" is reached where abundance is lower and species richness and biomass are higher. Biomass may be at a maximum in the "transition zone" because the organic enrichment of the sediments is enough to provide an abundant food source while not depleting the oxygen supply. At still greater distances the normal community is encountered (Pearson and Rosenberg, 1978). Diversity measured by the H' index typically reflects the above pattern (Gray, 1981) and is lowest at the "peak of opportunists". Dominance is generally at a maximum in the same area (Gray, 1981).

Evaluation of abundance data showed no increase in this factor at the near-ZID stations relative to other more distant stations for any

of the outfalls examined. Thus there is no indication of either the "peak of opportunists" or the somewhat elevated densities of the "transition zone" described by the Pearson and Rosenberg (1978) model. Similarly, the pattern for number of species showed no decrease in the ZID or near-ZID stations as would be typical for the "peak of opportunists" zone. Because distant stations could not be separated based on this parameter, there is also little evidence for the "transition zone" of the Pearson and Rosenberg (1978) model based on data on number of species. The lack of overall differences among stations is also suggested by cluster analysis. For all three study areas, all stations for an outfall survey were generally similar and there was little evidence of grouping based on proximity to the diffuser. Evidence from trends in both abundance and species richness for each of the three faunal fractions examined at the three outfalls would thus suggest that the bottom areas near these outfalls are not receiving sufficient organic material input to cause any aspects of the stressed commmunity responses described in the Pearson and Rosenberg (1978) model.

At the three Hawaiian ocean outfalls studied, there was no evidence of increased organic loads in the sediments at stations near the diffusers. No consistent pattern of percent total volatile solids in the sediments versus proximity of a station to the outfall diffuser was present. Sediment oxidation-reduction potential values (Nelson, 1986; Nelson et al., 1987a, 1987b) also indicated that the sediments were oxygenated. The measurements of these parameters agreed with qualitative inspection of the sediment samples which showed no evidence of reduced surface sediments even at stations immediately

adjacent to the diffuser pipes. Dollar (1986) estimated from work on nutrient fluxes at the Sand Island and Barbers Point Outfalls that only 1% of particulate effluent reaches the sediment surface at these locations, with the rest being dispersed by currents. Patterns of total volatile solids and oxidation-reduction potential at the Mokapu study area where dispersion would be expected to be even more rapid due to its more exposed location are consistent with those observed from the Barbers Point and Sand Island studies (Dollar, 1986; Nelson, 1986; Nelson et al., 1987a, 1987b).

Station B1 at the Mokapu study area was the only station within the ZID with clearly depressed abundance and species richness of all taxa versus other stations. Station B1 was deficient in fine grain size sediment (3-8 phi) relative to other Mokapu stations, having only 1.3% fine sediments versus a range of 5.0 to 13.2% at the other four stations. This station was also deficient in the number of sessile species, particularly polychaete species, which tend to be associated with more stable sediments possessing a higher percentage of fine sediment (Nelson et al., 1987). It is therefore probable that the decreased abundance of benthos at B1 was due to the sediment composition at this site and not to an effect of the diffuser effluent, although further sampling will be required to confirm this.

Based on the analysis of data from two Hawaiian outfalls, Nelson et al. (1987) suggested that the polychaete Pionosyllis gesae, the chordate Branchiostoma sp. A, and the gastropods Cerithidium perparvulum and Diala scopulorum may potentially be indicator organisms for outfalls in Hawaiian waters. With the addition of the data from Barbers Point, analysis of these individual species patterns

proved inconsistent as indicators to the proximity of an outfall diffuser. At present, it would appear that patterns of abundance for individual species are not a reliable indicator of outfall effects in Hawaiian waters. This may occur because such effects are minimal and the abundances of benthic organisms in these systems are primarily responding to other factors unrelated to the outfalls.

Acknowledgements

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A STUDY OF BEACH EXPOSURE FROM AN OFFSHORE MARINE OUTFALL

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ABSTRACT

An industrial outfall off coastal New Jersey was studied to determine the frequency and intensity at which local bathers might be exposed to diluted effluent. The data set was designed to provide exposure estimates for a comprehensive risk assessment requested by State regulatory agencies.

The effluent was tagged with rhodamine WT dye for a four month period from May to September 1986. The dye injection rate was continuously varied to maintain a concentration of 200 ppb at different effluent flow rates.

Four times per day, during the four-month study, grab samples were taken from the surf zone at stations along a four-mile stretch of beach centered at the location of the outfall. The samples were analyzed for fluorescence at rhodamine WT emission wavelengths.

Two ten-day plume mapping surveys were done to define the lateral extent of the diluted effluent. The plume generally tended southward parallel to the shoreline in a thin layer centered between 6 and 13m below the surface.

The fluorescence measurements were supplemented with continuous records of currents and tides and daily observations of vertical density stratification.

The data show that beach impingements occurred at a frequency close to a dominant frequency in the alongshore current spectrum. There is also qualitative evidence that the onshore/offshore movement of water is associated with vertical movement of the pycnocline.

BACKGROUND

The discharge permit for the Ciba-Geigy plant at Toms River, New Jersey was reissued in 1985. As a condition of the permit renewal, the company was required to monitor the local bathing beaches and conduct a fluorescent dye study to identify the extent of the effluent plume. The plant discharges approximately $0.3m^3/s$ of treated effluent into the Atlantic Ocean through a 305m diffuser located between 760 and 1,065m off Ortley Beach, New Jersey.

PRELIMINARY RISK ASSESSMENT

To select a meaningful discharge dye concentration, a preliminary risk assessment was developed by ENVIRON, INC. (ENVIRON, 1986). The intent of this assessment was to ensure that the dyed effluent could be detected at all dilutions that might represent a public health concern.

There were two fundamental unknowns in the assessment: the toxicity of the effluent and the degree of exposure. The Toms River plant manufactures dye stuffs and adjusts its product stream based on market factors. The composition of

the effluent therefore varies as, presumably, does the toxicity. Battelle Columbus (Battelle, 1986) has analyzed samples of the plant effluent and identified a number of the constituents. The concentrations vary, and there remains a number of unidentified components. Four priority pollutants have been found. All are nitrogen-containing organics. ENVIRON, therefore, chose to model the toxicity of the effluent using 0-dianisidine because it has the highest unit cancer risk (UCR) of all the nitrogen-containing organic compounds used in the manufacturing process. To be conservative, ENVIRON assumed a concentration of 5 parts per million (ppm) which is five times greater than the concentration of any nitrogen-containing compound identified by Battelle. It should be added that 0-dianisidine has not been found in the effluent.

The assumed exposure of swimmers was total immersion in the surf zone four days per week from May 15 through September 15 from age two through age seventy. An impingent was assumed to occur every five days at a dilution of 200 to 1. Using this scenario and conservative estimates of skin absorbtion and oral ingestion, the lifetime cancer risk was calculated to be 5.9×10^{-7} which is below New Jersey drinking water standards.

ENVIRON concluded that a discharge dye concentration that enabled measuring a 1000 to 1 dilution would be sufficient to assure that no potentially harmful levels of effluent could escape detection. The field study was designed in part to update the risk assessment by providing quantitative values for the frequency at which impingements occur and the dilution of the effluent in the surf zone during an impingement event.

STUDY DESIGN

There were three phases to the field study: beach monitoring, effluent plume mapping, and oceanographic data collection. The first two phases centered around the use of rhodamine WT dye as an effluent tracer.

Dye Dosing

The limit of detection of rhodamine WT in natural waters is controlled by the variance in background fluorescence at rhodamine emission wave lengths. This variance was unknown at the time the study was designed. To be conservative, it was assumed that a concentration of 0.2 parts per billion (ppb) could be reliably measured with an optical filter fluorometer such as the Turner Designs Model 10. This is

approximately an order of magnitude greater than might be expected further offshore. Therefore, to be sure of the ability of measuring a dilution of 1000 to 1 as specified by ENVIRON, the discharge dye concentration was set at 200 ppb.

Since the Ciba-Geigy effluent flow rate is adjusted to keep pace with manufacturing and treatment needs, it was necessary to make proportional adjustments in the dye injection rate to maintain a level discharge concentration. A variable speed dye injection pump was used which has a motor whose speed was controlled by a 4-20 milliamp signal from an inline flow meter used by the plant to monitor the discharge rate.

The discharge dye concentration was verified by sampling the dyed effluent downstream of the dye injection point and comparing the measured fluorescence with the fluorescence of a 200 ppb standard made from effluent collected upstream of the injection point. During the four months of the study (May 15 through September 15, 1986), the discharge concentration was within $\pm$ 10 percent of 200 ppb 66 percent of the time. Departures from the 200 ppb target were probably caused by a large quantity of entrained air bubbles which generated an

erratic response in the 4-20 milliamp signal. Nevertheless, the total rhodamine WT dosing was slightly greater than the amount that would have been required to give a constant 200 ppb over the four month injection period.

Beach Monitoring

The objective of the beach monitoring phase of the study was to determine the frequency, dilution, and locations at which effluent reaches the surf zone. Surf water samples were collected four times per day (0730, 1030, 1330, and 1630) from nine stations equally spaced north and south of the outfall (Figure 1). Immediately after the water samples were collected, they were taken to a laboratory area set up in Ortley Beach where the fluorescence was measured and converted to rhodamine WT concentration.

Water samples for background fluorescence were also collected daily at the control stations shown in Figure 1. This background water was used as the matrix for preparing the fluorometer calibration standards.

Plume Mapping

The distribution of rhodamine WT in the ocean was mapped for 11 consecutive days from May 21 through May 31 and for 10

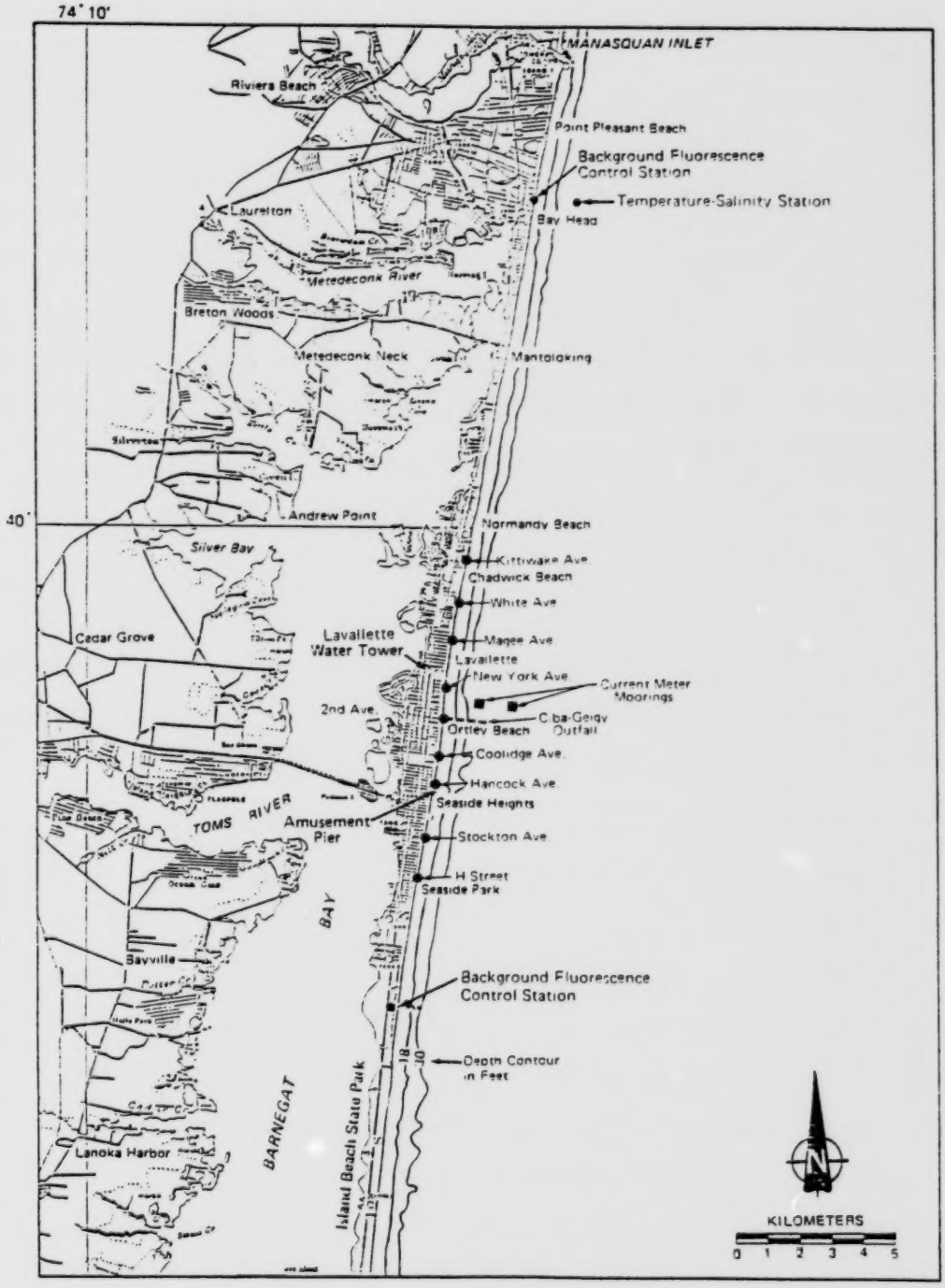

Figure 1. Site map showing locatons of beach monitoring stations, current meter mooring, background fluorescence control stations, and temperature/ salinity reference station.

consecutive days from August 1 through August 10. Dye measurements were made from the R/V Skimmer which was leased from the University of Delaware.

Water was pumped from the sampling depth through polyethylene tubing to a shipboard fluorometer. From the fluorometer the water passed through a temperature sensor, through the sampling pump, and then over the side. The sampling tube was attached to a weighted depressor which was towed alongside the boat with 6mm wire rope. The sampling depth was controlled by raising or lowering the depressor with a hydraulic winch.

During the May mapping period a single sampling tube was attached to the depressor. During the August mapping, three tubes were used to improve the vertical resolution and to avoid losing the plume as the depth of maximum fluorescence changed. The tubes were 1m apart and were connected to three individual fluorometers.

On arriving at the diffuser site, a series of vertical profiles was taken to locate the depth of the maximum rhodamine fluorescence. During the May mapping, the sampling tube was towed at that depth. During the August mapping, the middle of the three tubes was towed at that depth.

Oceanographic Data Collection

The oceanographic data consisted of current measurements, tidal elevation measurements, temperature/conductivity measurements and meteorological data acquisition. This data was collected to develop a qualitative understanding of the near shore hydrodynamics and how the various parameters might be related to the occurrence of effluent in the surf zone.

Two Endeco Model 174 current meters were moored at the location shown in Figure 1. The water depth was 16m and the meters were moored at 13m and 6m. There is a full four-month record for each meter. An Applied Microsystems in situ tide recorder was placed at the base of the current meter string, and that record was likewise complete.

Vertical profiles of temperature and conductivity were taken daily at the reference station shown in Figure 1. A Beckman RS-5 salinometer was used for temperature and conductivity. Salinity and density were computed in accordance with recommendations of the Joint Panel on Oceanographic Tables and Standards adopted in 1980.

Wind and atmospheric pressure data were provided by NOAA from the Atlantic City Airport.

RESULTS

Beach Monitoring

An impingement was deemed to have occurred when the rhodamine concentration, as determined by fluorescence measurements, exceeded 0.1 ppb (2000 to 1 dilution) in one or more surf water samples. There were 12 such events during the May 15 to September 15 study.

If it can be assumed that the 0730 to 1630 sampling period was not unique with respect to the probability of impingement, one can estimate by scaling that there would have been 32 impingements had the sampling been done around the clock. This implies that there was one impingement every 3.9 days. The highest rhodamine concentration was 0.16 ppb (1250 to 1 dilution). The preliminary risk assessment assumed exposure at 1.0 ppb (200 to 1 dilution) once every 5 days.

Figure 2 is a statistical summary of the beach monitoring data. Figure 2(a) shows that concentrations of 0.1 ppb or greater (2000 to 1 dilution) were present slightly less than 4 percent of the time. Figure 2(b) shows that impingements were more likely south of the diffuser.

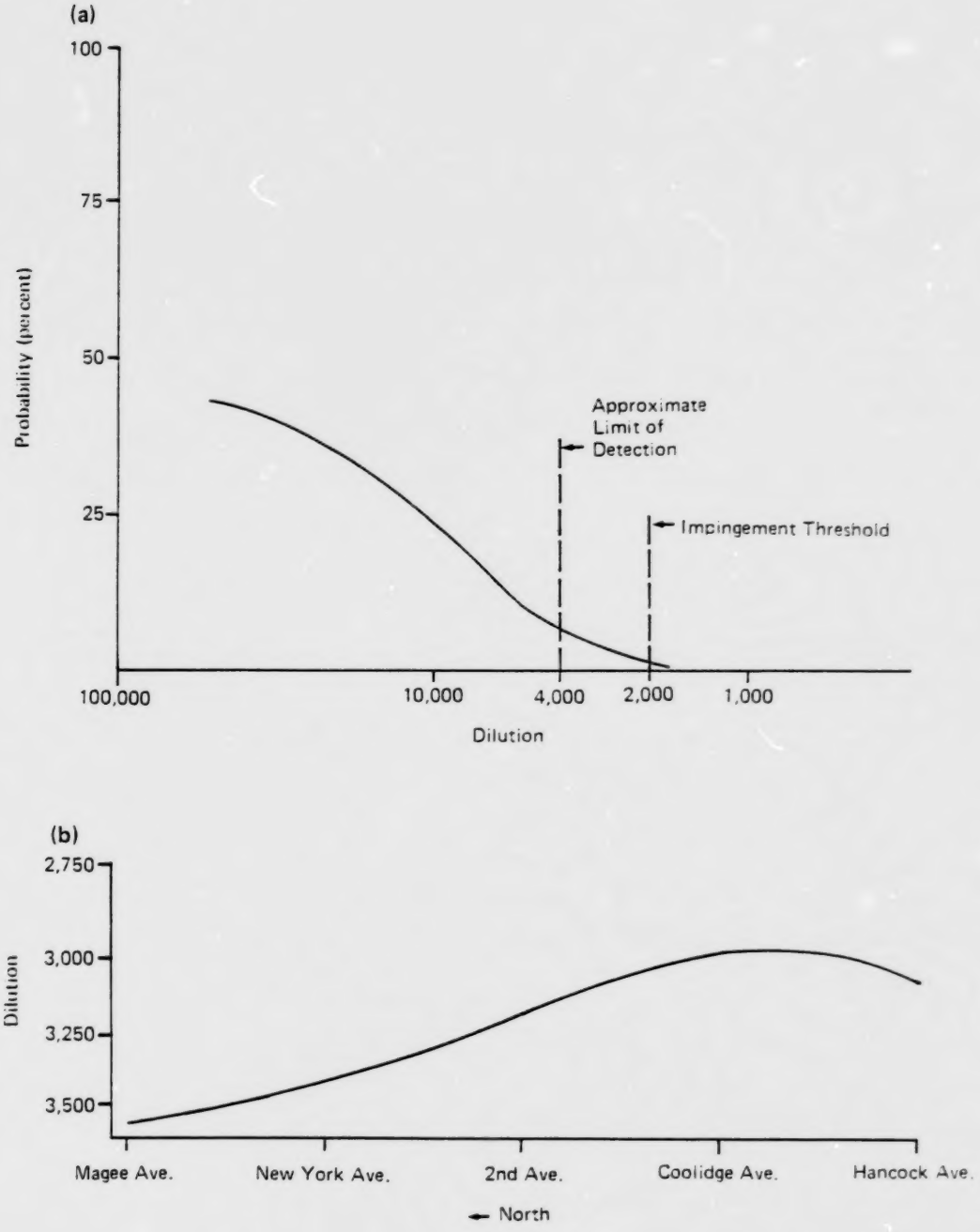

Figure 2. Statistical summary of beach monitoring data. (a) The probability of the presence of effluent at a given dilution in a water sample collected at any one of the nine beach monitoring stations; (b) the effluent dilution which was exceeded 95 percent of the time at the designated beach monitoring stations.

Plume Mapping

Figure 3 and Figure 4 show typical dilution patterns for the May and August surveys respectively. The contours were developed from dye concentration data which was collected over two to three hours and should not be interpreted as synoptic. It should be noted that when sampling at a single depth such as in May, it would be possible to move into and out of the region of maximum fluorescence if the plume rose or sank in response to local variations in the vertical density structure. Thus, some of the dilution contours in Figure 3 might have been larger or smaller had the sampling depth been adjusted up or down. This problem led to the decision to sample at three depths during the August survey.

The plume was found to be confined to a relatively thin layer centered between 6 and 13m below the surface. This is consistent with simple diffuser theory for linearly stratified conditions (Fischer et al., 1979). The height of rise is described by the following equation:

$$h = 2.8\ B^{1/3}\ (ge)^{-1/2}$$

where

h = height of rise

B = buoyancy flux per unit length of diffuser

g = gravitational acceleration

e = density gradient relative to effluent density

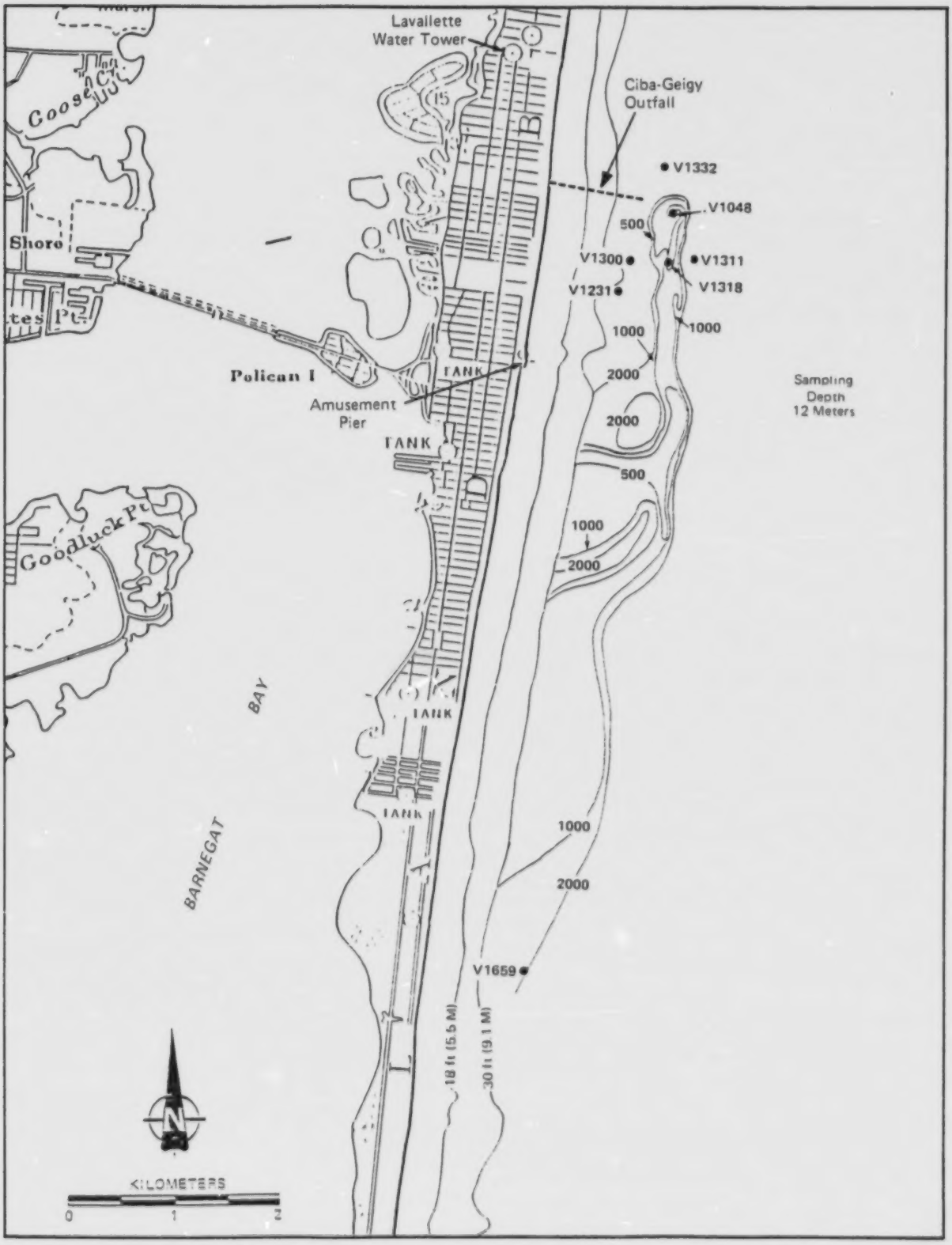

Figure 3. Contours of constant dye dilution at a depth of 12m on May 18, 1986.

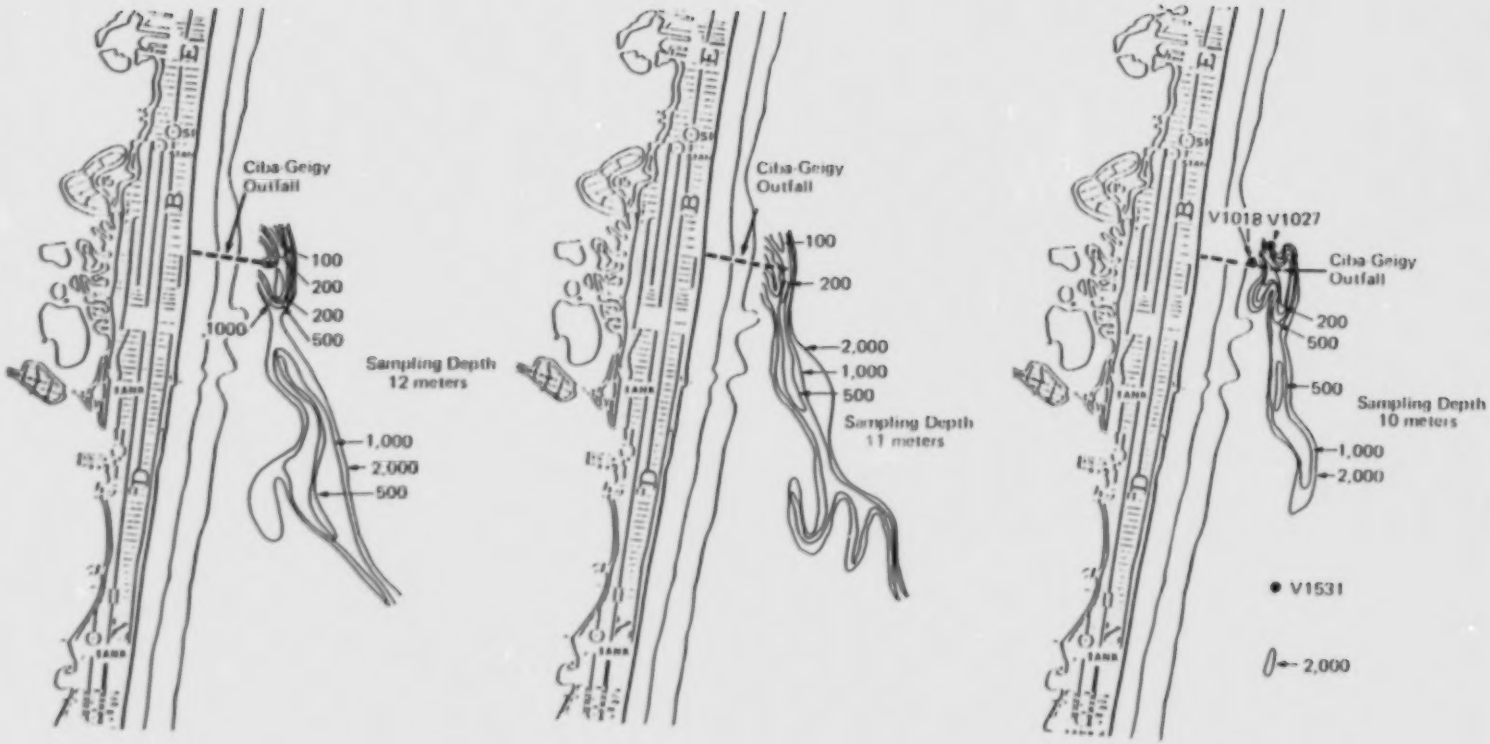

Figure 4. Contours of constant dye dilution at indicated sampling depths on August 4, 1986.

The computed height of rise for a $0.2m^3/s$ flow at a depth of 14m and a bottom to surface density difference of 1.5 sigma-T units is 4.3m. This would place the plume at a depth of 10m which generally agrees with the observations.

Oceanographic Data

The first step in analyzing the current data was calculation of the principal axes of the current vectors and computation of components parallel and normal to these axes. The principal axis is assumed to be in the alongshore direction. This, in effect, calibrates the internal compass in the meters using the argument that nearshore currents must, on the average, be parallel to the coastline. The component records were then filtered to remove constituents with periods less than 60 hours (Pritchard et al. 1984).

The tidal elevation record was corrected for atmospheric pressure and then passed through the same numerical filter as the current records.

Wind data from the Atlantic City Airport was obtained through NOAA. The 030^{o} component showed the greatest correlation with the alongshore currents and was used in the data comparison.

The density data is plotted in Figure 5 as contours of constant sigma-T versus depth and time. The vertical dashed lines represent times of beach impingement, and it is evident from the figure that effluent reached the beach during periods when the water column was stratified. Apparently, mechanisms other than vertical mixing are involved. The important aspect of Figure 5 is the correlation between impingements and an upward movement of the isopycnal lines.

Tide, current, wind, and beach monitoring results are all shown on Figure 6. The plots are synchronized in time so that the relationship between the records can be observed more easily. Table 1 lists the coherency and phase relationships corresponding to specific spectral peaks in the current, wind, and tide records. Of particular interest is the high coherence at 3.6 days between the alongshore current and wind and the high coherence at 3.5 days between the current and tide. At that period, the current and tide are almost 180 degrees out of phase which is consistent with classical coastal dynamics where a southerly alongshore current will tend to deflect to the right under the influence of the earth's rotation causing high water at the shore.

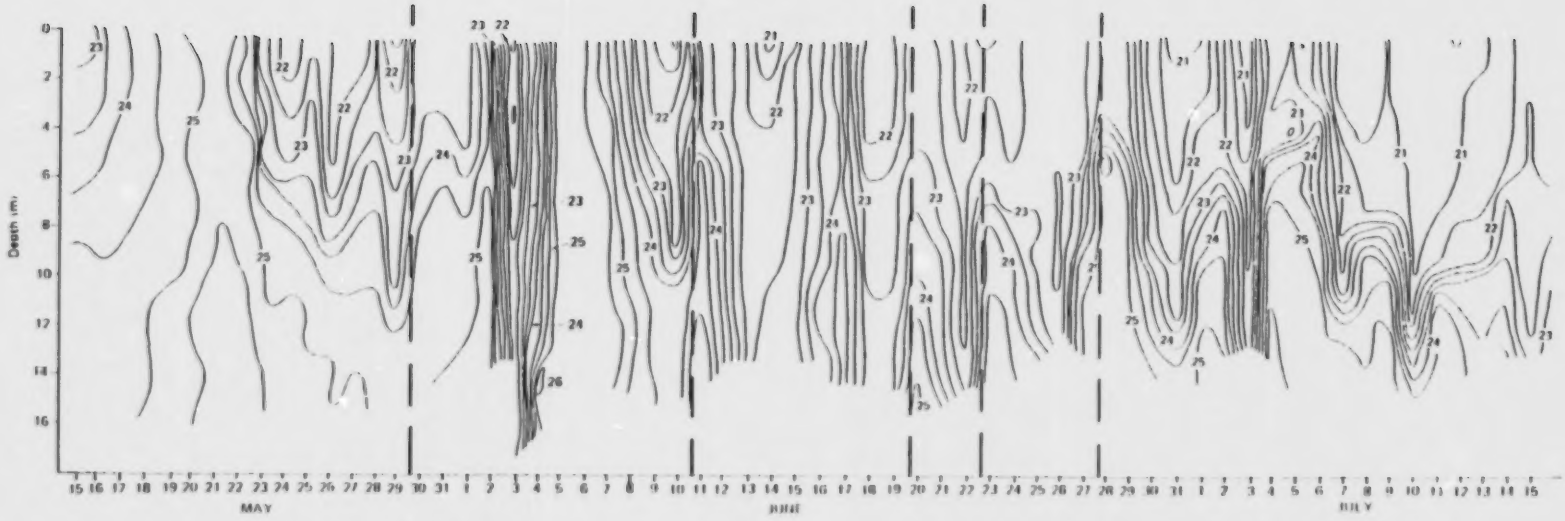

Figure 5. Contours of constant sigma-T versus depth and time for the period May 15-September 15, 1986 at Bay Head, New Jersey. Vertical dashed lines represent times of beach impingement.

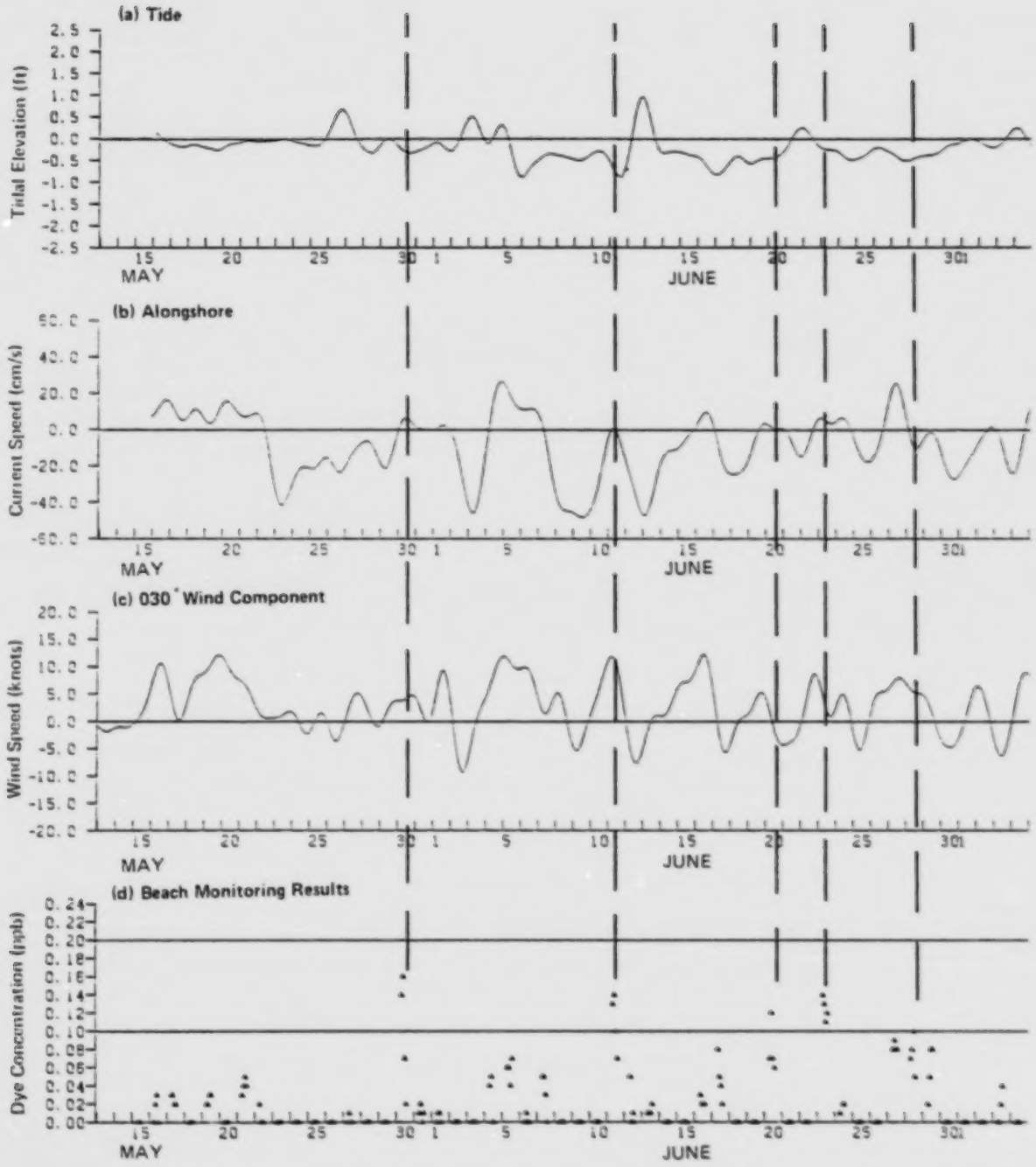

Figure 6. Filtered constituents of (a) tide, (b) alongshore current, (c) 030° wind component, and (d) beach monitoring results. Vertical dashed lines represent times of beach impingement.

TABLE 1 COHERENCY BETWEEN THE ALONGSHORE CURRENT AND THE 030 WIND COMPONENT; AND COHERENCY BETWEEN THE ALONGSHORE CURRENT AND TIDAL ELEVATION

Current/Wind

Frequency (day^{-1})	Period (days)	Coherence	Phase (hours)	95% Confidence Interval
0.098	10.2	0.96	4.1	0.95-0.97
0.131	7.6	0.79	15.2	0.77-0.81
0.172	5.8	0.87	10.1	0.86-0.88
0.221	4.5	0.68	4.8	0.65-0.70
0.279	3.6	0.94	9.8	0.93-0.95

Current/Tide

Frequency (day^{-1})	Period (days)	Coherence	Phase (hours)	95% Confidence Interval
0.107	9.3	0.51	96.7	0.48-0.54
0.147	6.8	0.61	80.2	0.58-0.64
0.189	5.3	0.66	-53.4	0.63-0.69
0.238	4.2	0.90	-47.9	0.89-0.91
0.287	3.5	0.86	-38.7	0.85-0.87

DESCRIPTIVE MODEL

A seasonal thermocline develops in the coastal ocean in the spring of each year and persists through the summer months before dissipating in the fall.

The vertical density structure can be modeled as a simplistic two-layer system consisting of a lighter surface layer overlying a heavier bottom layer and separated by a frictionless interface at the depth of the thermocline (Csanady, 1975). The "frictionless" assumption decouples the two layers, allowing the surface layer to move independently of the bottom layer.

The width of the coastal zone where this descriptive model applies is termed the internal radius of deformation. It is the ratio of the speed of propagation of internal waves along the pycnocline to the inertial frequency for the local latitude. For coastal New Jersey, the radius of deformation is approximately 5km.

If the coastline is oriented north and south, a surface layer current moving toward the south will tend to move onshore with accompanying downwelling and offshore movement in the lower layer. This depresses the pycnocline. If the current

in the surface layer slackens or turns to the north, the tilt of the pycnocline will be relaxed. An observer in the ocean will see the vertical temperature and density structure move up and down in time as the pycnocline tilt continually adjusts to changes in the surface current. A typical pycnocline slope is approximately 1 m in 1 km (Csanady, 1975). For a coastal zone 5 km wide, the amplitude of the vertical movement of the pycnocline near the beach could be as much as 5 m. In Figure 5, the amplitude is approximately 3 meters and seems to be closely associated with impingement events.

There is a large onshore/offshore transport of water associated with these changes in tilt, and this can be an important mechanism in flushing of coastal water.

CONCLUSIONS

The coastal dynamics in the study area appear to be dominated by changes in depth of the pycnocline caused by wind-driven changes in the alongshore current. An understanding of this process is important in evaluating the capacity of the nearshore zone to receive wastes.

A coupling of a beach monitoring program, an effluent characterization program, and relevant physical oceanographic measurements provided the basis for a conceptual model predicting swimmer exposure to a marine industrial discharge. For this ocean discharge, the present study demonstrated that the predicted frequency and concentration of waste effluent exposure to swimmers would not result in any unacceptable health risk.

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EFFECTS OF SEWAGE SLUDGE DISPOSAL IN THE NORTHWEST ARABIAN GULF REGION

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Surface sediment samples from the extreme N.W. Arabian Gulf region were analysed for its organic carbon, kjeldahl nitrogen, total phosphorous, with simultaneous determination of pH and Eh as a function of surface distribution of nutrients to identify the effects of the municipal and industrial waste water discharge into the semi-enclosed bay. The results of the study revealed that the discharge of sludge products into the bay stands as the dominant factor behind the anaerobic condition at the sediment-water interface and it also gives rise to enhancement of nutrient concentrations organic carbon and kjeldahl nitrogen in the sediments. The C/N ratio and the concentration of the elements decrease seaward and the controlling mechanisms for these are discussed. In general, the sediments show reliable indication of coastal nutrient enrichment from industrial discharges and the observed relationship of nutrients to organic carbon stands as an excellent measure of the nutrient load to this coastal environment.

MANUSCRIPT NOT AVAILABLE AT TIME OF PUBLICATION

TITLE: OVERVIEW OF THE COOPERATIVE INTERAGENCY (CE/EPA) TESTING OF DREDGED MATERIAL PROGRAM - FIELD VERIFICATION PROGRAM

BY ROBERT L. LAZOR, US ARMY ENGINEER WATERWAYS EXPERIMENT STATION

1.0 INTRODUCTION. Evaluation of proposed dredged material discharges under Section 103 of the Ocean Dumping Act and Section 404 of the Clean Water Act requires field-verified, state-of-the-practice procedures for predicting and assessing environmental effects. The Field Verification Program (FVP) is a cooperative effort between the Corps of Engineers (CE) and the Environmental Protection Agency (EPA) to field verify various testing procedures for implementing Section 404 and 103 requirements. Promising evaluative procedures were applied to a maintenance dredging operation near New Haven, Connecticut. Here the impacts of a highly contaminated sediment placed in upland, wetland and aquatic disposal environments were tested. The FVP is a 6-year program initiated in 1982.

2.0 OBJECTIVES. The three objectives of the FVP are to: (1) document in the laboratory predictive methods for assessing the effects of disposal of contaminated dredged material, (2) verify these methods in the field following disposal, and (3) conduct a comparative assessment of dredged material disposal in aquatic, wetland, and confined upland disposal environments.

3.0 PROJECT DESCRIPTION. The dredging site was Black Rock Harbor (BRH), located in Bridgeport, Connecticut, where maintenance dredging provided a channel 46 m wide and 5.2 m deep at mean low water (mlw) (Figure 1). Approximately 55,000 m^3 of material was dredged during April and May 1983 and deposited in 20 m of water in the northeastern corner of the Central Long Island Sound (CLIS). The CLIS is a historical open water dredged material disposal site (1.8 by 3.7 km), located approximately 15 km southeast of New Haven, Connecticut (Figure 1). Bathymetric profiles associated with the CLIS open water site are provided in Figure 2. The sediment was clam-shell dredged and deposited with bottom-opening barges.

3.1 Confined upland/wetland disposal sites were constructed at Tongue's Point, Connecticut adjacent to Bridgeport Harbor, approximately from the Black Rock Harbor Channel (Figure 3). The upland/wetland sites were created by grading and dike construction along the desired alignments (Figure 3). Material was excavated from the area planned for the wetland to allow placing dredged material at elevations suitable for wetland substrate. The excavated material was used to construct the upland dikes to contain dredged material at elevations suitable for upland disposal area.

3.2 The upland site has a surface area of approximately 2,418 m^2 within the inside dike toes. The bottom of the site was graded to elevation plus 2.13 mlw. The depth of the dredged material was approximately 0.91 m following disposal, sedimentation, and consolidation. The wetland site had a surface area of approximately 744 m^2 within the inside excavation toes. The bottom of the site was graded to approximately elevation plus 0.91 m mlw. The depth of the dredged material fill was also 0.91 m following sedimentation and consolidation.

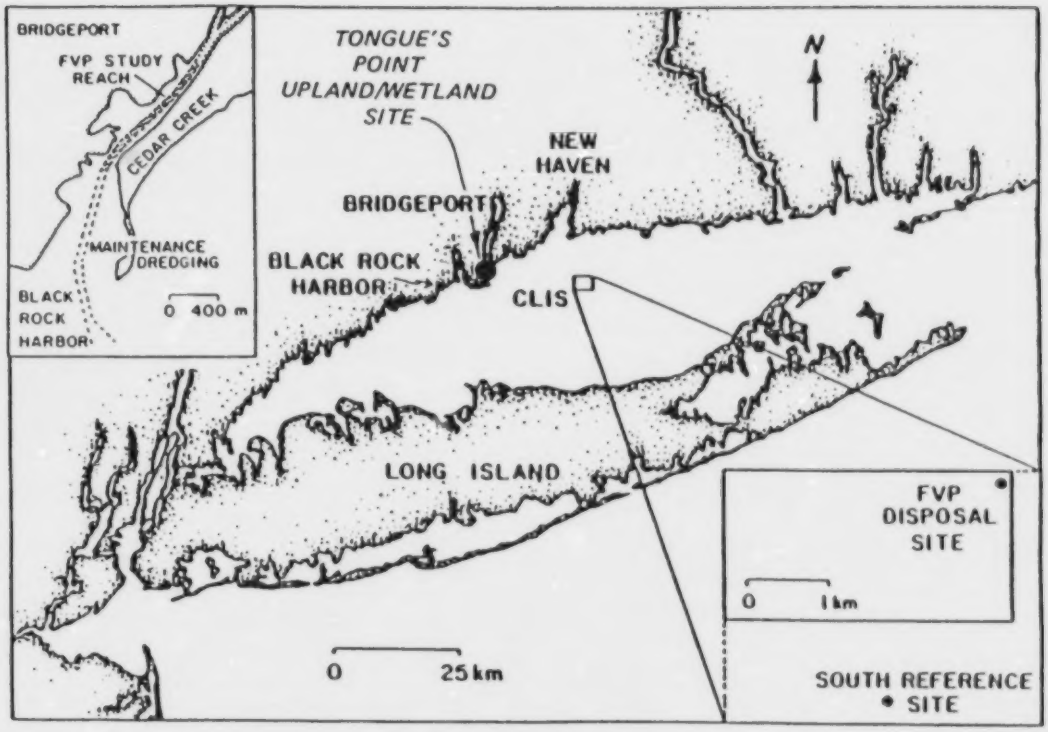

Figure 1. Central Long Island Sound (CLIS) disposal site and Black Rock Harbor (BRH) dredge site

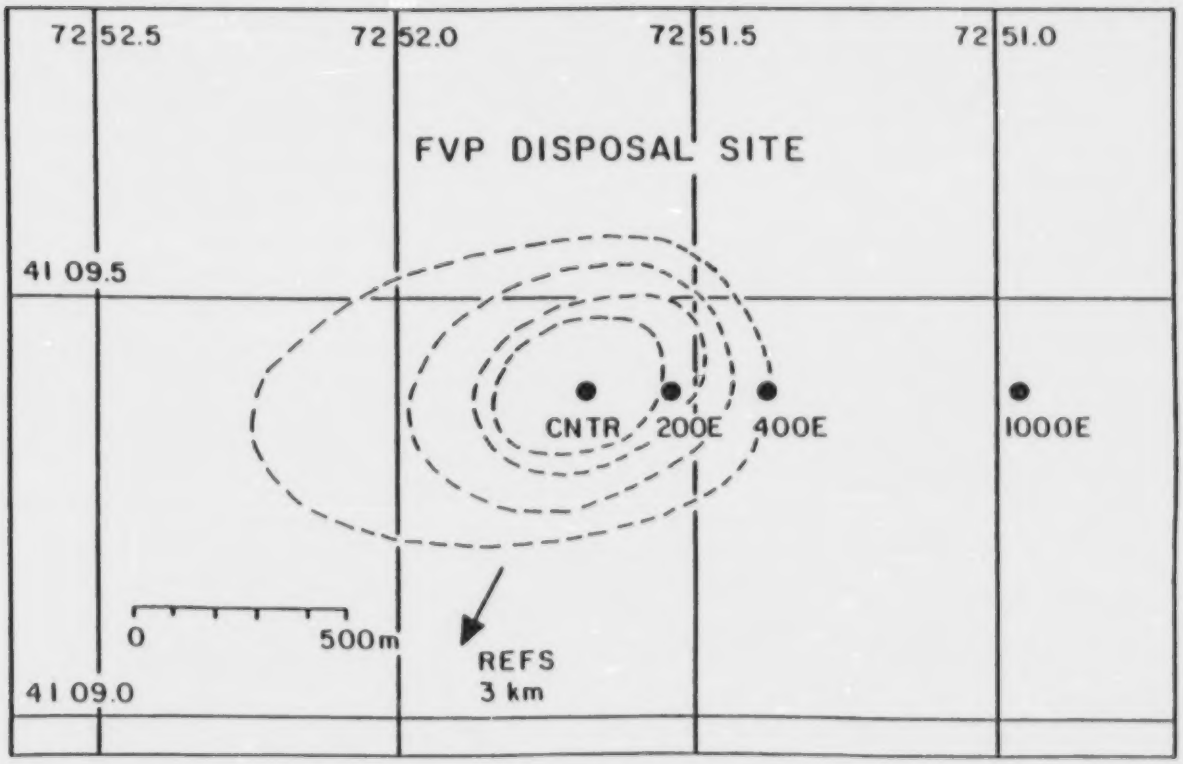
72 52.5
72 52.0
72 51.5
72 51.0
FVP DISPOSAL SITE
41 09.5
CNTR
200E
400E
1000E
0
500m
REFS
3 km
41 09.0

Figure 2. FVP

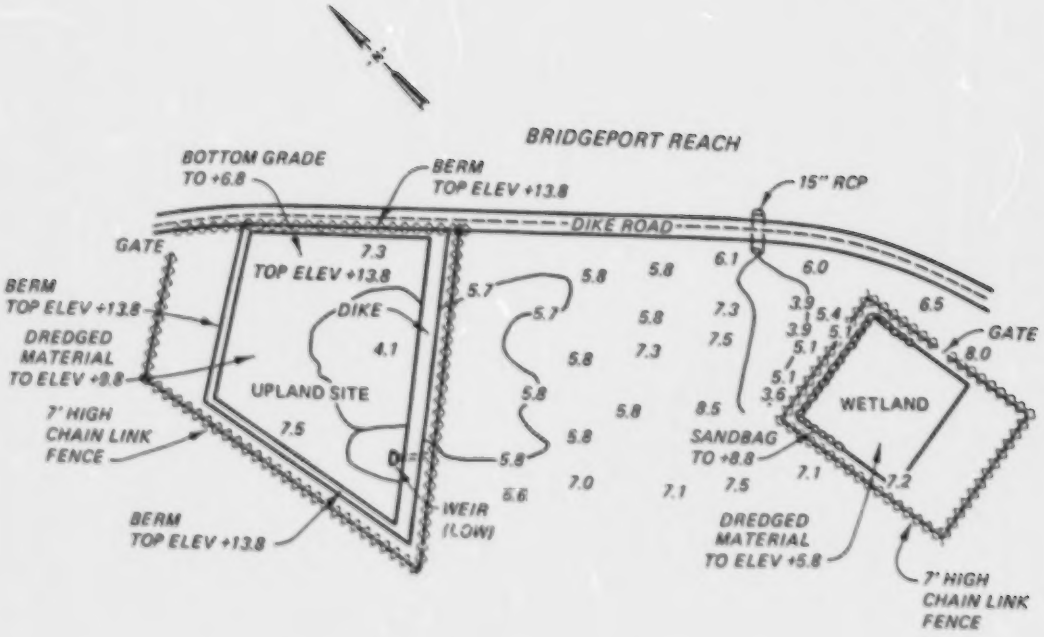

Figure 3. Constructed FVP upland/wetland sites

4.2.1 Effects of upland disposal on water quality. Laboratory tests for predicting effluent quality were conducted on BRH sediments prior to placement in the confined disposal area. The confined disposal area was designed, operated, and managed to ensure optimum fill configuration for the field studies and evaluation of water quality effects. During filling operations water quality parameters were monitored extensively for influent and effluent and at selected stations within the disposal area. Following disposal, the quality of surface water runoff was determined by controlled simulation of rainfall and collection of surface water samples. Monitoring wells were placed around and within the disposal area, and groundwater samples collected prior, during and after filling.

4.2.2 Bioaccumulation of contaminants in upland and wetland plants. First generation test procedures were verified at the field site, and laboratory bioassay tests were conducted under wetland and upland conditions. Field bioassay tests were conducted to verify laboratory test results.

4.2.3 Bioaccumulation of contaminants in upland and wetland animals. Existing upland and wetland laboratory animal bioassay test results were field tested for verification. Laboratory procedures developed in Europe included those using selected upland and wetland animals.

5.0 FVP PROGRAM SYNTHESIS. Results of the aquatic, wetland and upland studies will be synthesized in a Field Verification Program Synthesis Report and will provide a comparative assessment of effects associated with the various disposal options. Dr. Tom Dillon (CE) reported in detail on this aspect of the FVP in these proceedings.

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APPLICABILITY AND FIELD VERIFICATION OF PREDICTIVE METHODOLOGIES FOR AQUATIC DREDGED MATERIAL DISPOSAL

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ABSTRACT

The Field Verification Program (FVP) was designed to: (a) determine the applicability, degree of variability, and reproducibility of existing test methods to detect and measure effects of dredged material; (b) to field verify laboratory responses by comparing the exposure-response relationships between the laboratory and field; and (c) to determine the degree of correlation between contaminant tissue residues and biological responses. These objectives were examined for the following biological responses: bioaccumulation, scope for growth (SFG), bioenergetics, adenylate energy charge (AEC), sister chromatid exchange (SCE), histopathology, survival, growth, reproduction, intrinsic rate of population growth, recolonization and community structure.

The following recommendations are made regarding the application of the test methods in this study for dredged material evaluation: (a) measures of survival, growth, reproduction, population growth rate, scope for growth, bioenergetics and contaminant bioaccumulation are recommended for the predisposal evaluation of dredged material; (b) while a genotoxic test method is desirable in predisposal evaluations, sister chromatid exchange is not recommended pending further development of the test method; (c) acceptable methods for postdisposal field assessments include scope for growth, growth, and bioaccumulation measured in *M. edulis* and bioenergetics and bioaccumulation in *N. incisa*; (d) REMOTS is recommended for rapid reconnaisance and characterization of benthic communities and physicochemical properties of the sediments. Species classification and enumeration techniques are recommended for definitive community assessments.

Examination of dredged material fate indicated that 70-80% of the material and 30-60% of selected contaminants reached the disposal site. Near-bottom water column contaminant transport was detected during and for one month post-disposal. Surficial sediment (0-2 cm) contaminants on the mound decreased significantly after nine months in contrast to deeper sediments that remained a source of contamination for the three year study period. Biological effects (SFG) were detectable in the near-bottom water during the one month post-disposal in the vicinity of the mound. Bioenergetics effects in infauna on the mound persisted for three years. Contaminant tissue residues for PCBs and the higher molecular weight PAHs remained elevated in infauna for one to three years as a function of distance from the disposal mound.

Recolonization and community succession were indistinguishable from the reference site within nine months of disposal 400 and 1000 meters from the mound apex but did not return to reference conditions after three years within a 200 meter radius of the mound apex.

BACKGROUND

The Marine Protection, Research, and Sanctuaries Act (Public Law 92-532) was passed by Congress in 1972. This law states that it is the policy of the United States to regulate disposal of all types of materials into ocean waters and to prevent or strictly limit disposal of any material that would adversely affect human health or welfare, the marine environment, or ecological systems. The implementation of this law, through the issuance of permits as defined in the final regulations and criteria, is shared jointly by the US Environmental Protection Agency (USEPA) and the US Army Corps of Engineers (CE).

In 1977, the CE and the USEPA prepared technical guidance for the implementation of the final ocean dumping regulations in the form of a manual entitled "Ecological Evaluation of Proposed Discharge of Dredged Material into Ocean Waters" (USEPA/CE 1977). This manual specified which test procedures were to be followed in collecting information to be used in making a disposal decision. Among the procedures were those for: (a) chemically characterizing the proposed dredged material; (b) determining the acute toxicity of liquid, suspended particulate, and solid phases; (c) estimating the potential contaminant bioaccumulation; and (d) describing the initial mixing during disposal. These methods have been used for determining the suitability of dredged material for open-water disposal. The procedures in this manual represented the technical state of the art at that time and were never intended to be inflexible methodologies. The recommended test methods were chosen to provide technical information that was consistent with the criteria specified in the regulations. However, use of the manual in the permit process has identified both conceptual and technical limitations with the currently recommended test methods (Gentile and Scott 1986).

To address the limitations of the testing program, the Interagency Field Verification of Testing and Predictive Methodologies for Dredged Material Disposal Alternatives Program or the Field Verification Program (FVP) was authorized in 1982. This 6-year program is sponsored by the Office, Chief of Engineers, and is assigned to the US Army Engineer Waterways Experiment Station (WES), Vicksburg, Miss. The objective of this interagency program is to field verify existing test methodologies for predicting the environmental consequences of dredged material disposal under aquatic, wetland, and upland conditions. The aquatic portion of the FVP was conducted by the USEPA Environmental Research Laboratory, Narragansett, R. I. (ERLN). The wetland and upland portions, conducted by WES, will be reported in separate documentation.

PROJECT DESCRIPTION

The disposal site for the FVP is an historical site known as the

Central Long Island Sound (CLIS) disposal site (1.8 by 3.7 km) located approximately 15 km southeast of New Haven, Conn. (Figure 1). The sedimentology at the disposal and reference sites is primarily silt-clay, with a mean grain size of 0.013 mm. Thermal stratification occurs from April to September, and during this period, bottom salinity is slightly higher than that of the surface. Tidal currents typically dominate the near-bottom water in an east-west direction. Suspended sediment concentrations average 10 mg/l, with storm-induced values approaching 30 mg/l, measured one meter above the bottom. The baseline biological community data revealed a homogeneous, mature infaunal community dominated by the polychaete Nephtys incisa and the bivalve molluscs Nucula proxima and Yoldia limatula.

The FVP disposal site was selected within the CLIS so as to minimize contamination from other sources, including relic disposal operations and ongoing disposal activities occurring during the study period. This was necessary to ensure a point source of contamination for the study. The uniformity of physical, chemical, and biological properties of the disposal site prior to disposal allowed detection of changes in these properties due to the disposal of the dredged material. Finally, the stations used to study the biological effects in this study were selected along the primary axis of total current flow to represent a gradient of potential exposure for the biota (Figure 2).

The spatial scale of this study is near-field and limited to the immediate vicinity of the disposal site. A primary assumption is that the mound of dredged material constitutes a point source of contamination. The temporal scale for the study is 4 years, which includes a year of predisposal data collection to define seasonal patterns in the physical, chemical, and biological variables and 3 years of post-disposal data collection to address the objectives of the program and to evaluate the long-term impacts of the disposal operation on the surrounding benthic communities.

The dredging site was Black Rock Harbor (BRH), located in Bridgeport, Conn., where maintenance dredging provided a channel 46 m wide and 5.2 m deep at mean low water (Figure 1). Approximately 55,000 m^3 of material were dredged during April and May 1983 and disposed in 20 m of water in the northeastern corner of the CLIS disposal site.

The dredged material from Black Rock Harbor contained substantial concentrations of both organic and inorganic contaminants. A subset of the total contaminant profile examined in this project is presented in Table 1. Polychlorinated biphenyls (PCBs) were present in the dredged material at a concentration of 6,400 ng/g, and polynuclear aromatic hydrocarbons (PAHs) with molecular weights between 166 and 302 were present at concentrations ranging from 1,000 to 12,000 ng/g. Alkyl homologs of the PAHs were also present in the dredged material at concentrations between 1,000 and 13,000 ng/g. Inorganic contaminants of toxicological importance present in the dredged material include copper (2,900 ug/g), chromium (1,480 ug/g), zinc (1,200 ug/g), lead (380 ug/g), nickel (140 ug/g), cadmium (24 ug/g), and mercury (1.7 ug/g).

Table 1

Concentrations of the Ten Selected Contaminants and Two Summary Statistics for Both BRH and REF Sediments. (Means ± Standard Deviations)

Chemical Compound	Sediment* BRH	REF
Phenanthrene	5,000 ± 1,800 (15)**	85 ± 17 (12)
Sum of 178 alkyl Homologs	28,000 ± 8,300 (15)	170 ± 26 (12)
Fluoranthene	6,300 ± 1,300 (15)	240 ± 33 (12)
Benzo(a)pyrene	3,900 ± 970 (15)	250 ± 28 (12)
Ethylan	4,000 ± 820 (15)	0 ± - (12)
PCB as A1254	6,400 ± 840 (15)	39 ± 4 (12)
Sum of PAHs	142,000 ± 30,000 (15)	4,500 ± 510 (12)
Centroid of PAHs	232.8 ± 1.7 (15)	249.2 ± 1.7 (12)
Copper	2,900 ± 310 (18)	60 ± 3 (15)
Cadmium	24 ± 0.6 (18)	0.23 ± 0.04 (15)
Chromium	1,480 ± 83 (18)	50 ± 15 (15)
Iron	31,000 ± 2,800 (18)	21,000 ± 1,400 (15)

* Units are nannograms/gram dry weight for the organic compounds and the statistic Sum, micrograms/gram dry weight for the inorganic elements, and molecular weight for the statistic Centroid.

** (N) = number of sediment samples analyzed.

DISCUSSION

The Field Verification Program (FVP) was designed to test the applicability, reproducibility, and verification of biological test methods to be used in the evaluation of dredged material impacts in aquatic, wetland and upland disposal sites. Three components comprise the aquatic portion of the FVP: (a) laboratory documentation, (b) field verification, and (c) resi-

due effects relationships. The comparisons of the three disposal options will be discussed in the FVP program synthesis report. The details and results of the research summarized in this report are the subject of fifteen published technical reports. The following summarizes the results from the aquatic portion of the FVP and provides a series of recommendations on the utility of the test methods.

Laboratory Documentation

The objective of these studies was to determine if the test methods could quantitatively and predictably describe biological responses to dredged material exposure in a variety of marine species and to determine the variability and reproducibility of the response to replicate exposures of dredged material. Laboratory test systems were developed to provide continuous, controlled exposure of suspended and bedded sediments to eleven species for acute toxicity studies reported in Rogerson et al. (1985) and four species were examined in chronic toxicity studies. All the biological responses measured in the four principal marine test organisms exposed to bedded and/or suspended sediments in the laboratory are summarized below and in Table 2.

- The ability of the biological test methods to respond quantitatively to varying exposures of dredged material was demonstrated for all responses except adenylate energy charge in M. edulis and N. incisa and sister chromatid exchange, and histopathological responses in N. incisa.

- The reproducibility of the test methods was assessed in the laboratory through replicate tests with dredged material. Only adenylate energy charge, sister chromatid exchange, and histopathology did not replicate in a quantitatively acceptable manner. Histopathological responses were qualitatively reproducible in that similar lesions occured in the same species repeatedly but the response frequency was not reproducible.

The laboratory documentation phase of the program demonstrated that biological test methods could be used to reproducibly measure the effects of dredged material in the laboratory. Having applied these methods in the laboratory, the next phase of the program was to evaluate the test methods in the field and compare the results with laboratory predictions.

Field Verification

The objective of the field verification component of this program was to verify the laboratory test methods and predictions in the field. Stated as a null hypothesis, the exposure-response relationship developed in the laboratory is not significantly different from a similar relationship in the field. To test this hypothesis it is necessary to describe the field exposure conditions to the same level of detail as is done in the laboratory. Continuous, comprehensive time-variable field exposure information was not acquired in this

Table 2

Summary of Results for Assessing the Success and Applicability of Test Methods for Evaluating Dredged Material Effects

Biological Response	Laboratory Documentation				Regulatory Application	
	Exposure-Response Relationship	Replicate Reproducibility of Test Method	Field Verification	Residue Effects	Pre-Disposal Evaluation	Monitoring Application
Bioaccumulation						
M. edulis	+	+	+	NA	+	+
N. incisa	+	+	+	NA	+	+
Scope for Growth						
M. edulis	+	+	+	+	+	+
Bioenergetics						
N. incisa	+	+	+	+	-	+
AEC						
M. edulis	-	-	+	+	-	-
N. incisa	-	-	+	+	-	-
SCE						
N. incisa	-	-	+	+	-	-
Histopathology						
M. edulis	+	+	-	+	-	-
N. incisa	-	-	-	-	-	+
Mortality						
M. bahia	+	+	NA	NA	+	-
A. abdita	+	+	NA	NA	+	+

(Continued)

Table 2 (Continued)

Growth						
M. edulis	+	+	+	+	-	+
N. incisa	+	+	NA	NA	+	-
M. bahia	+	+	NA	NA	+	-
A. abdita	+	+	NA	NA	+	-
Reproduction						
M. bahia	+	+	NA	NA	+	-
A. abdita	+	+	NA	NA	+	-
Population Growth						
M. bahia	+	+	NA	NA	+	-
A. abdita	+	+	NA	NA	+	-
Community						
Benthic	NA	NA	NA	NA	-	+
REMOTS	NA	NA	NA	NA	+	+

NA = Not applicable.

\+ Successful.

\- Unsuccessful.

study. Consequently, field exposures were estimated from discrete data that limited the degree of quantification for the field verification. Therefore, rather than attempt to precisely define the exposure field, upper and lower boundries were estimated and used to compare with the laboratory exposures. The results are as follows:

- The test methods for measuring bioaccumulation, scope for growth, bioenergetics, adenylate energy charge, sister chromatid exchange, and histopathology were successfully applied to the same species in both the laboratory and field. However, adenylate energy charge and sister chromatid exchange did not respond quantitatively to BRH exposure.

- There was excellent laboratory and field concurrence with the bioaccumulation of PCBs and PAHs, their chemical patterns, and the magnitude of accumulation in both M. edulis and N. incisa (Figure 3,4).

- Scope for growth in M. edulis showed concurrence between the laboratory and field when analogous exposure conditions were compared (Figure 5). Significant changes in field scope for growth measurements in M. edulis were limited to within two months after disposal and reflect the transient nature of the water column exposure. The bioenergetics responses of field populations of N. incisa which were exposed via the sediments continued to show statistically significant changes two years after disposal. In addition, there was concurrence between laboratory and field measures of respiration and excretion in N. incisa.

- Adenylate energy charge and related adenylate nucleotides were measured in M. edulis and N. incisa in both the laboratory and field. These responses, with the exception of the adenine pool, did not show a quantitative exposure-response relationship. The adenylate analysis in field N. incisa were complicated by technical problems with the measurement of the adenine nucleotides from field samples during the first three months after disposal. Considering these limitations, it is not surprising that attempts to correlate laboratory and field responses were unsuccessful.

- Sister chromatid exchanges, measured in N. incisa, were determined in both the laboratory and the field. The laboratory response, while present, lacked a predictable exposure-response relationship and did not replicate satisfactorily. An increase in SCE exchanges was detected in field samples collected during one post-disposal sampling period. Thus, the degree of concurrence between laboratory and field is essentially a qualititative one in that an increase in the SCE response was detected in both lab and field in the presence of BRH sediment.

- Histopathological effects were measured in M. edulis and N. incisa in both the laboratory and field. Concurrance between laboratory and field results for G.I. tract pathology in M. edulis was acceptable while the lack of concurrance for gill pathology was unexpected based upon laboratory results (Figures 6,7). A possible explanation for this discrepancy is the difference between the total integrated exposures in the laboratory and field. Significant histological change, epidermal thickening of the parapodis, in N. incisa was detected at only the highest laboratory exposure which was outside the range of field exposure. Within the overlapping range of laboratory and field exposures no significant histological change was detected.

- Benthic recolonization and community structure were determined by the identification and ennumeration of quantitative grab samples and the application of a remote reconnaisance technique (REMOTS) which integrates redox, granulometry, and biological interrelationships. The results indicated that the mature Stage III successional community had not returned after three years to areas of the disposal mound that had >10cm of deposited dredged material (Figure 8).

The above summarizes the field verification of selected laboratory responses. It is imperative that laboratory-field comparisons be made for analagous exposures in order to properly interpret the biological data. It is not surprising that those responses that did not perform well under laboratory conditions were the same ones that could not be verified in the field. Based upon the data in this study we accept the null hypothesis that exposure-response relationships developed in the laboratory are not qualitatively or quantitatively different from those in the field.

Residue - Effects Relationships

The purpose of this component of the study was to determine if there were correlations between specific contaminant tissue residues and observed biological responses. Bioaccumulation data provided evidence that contaminants were biologically available. This is of fundamental importance to establishing exposure patterns and supporting those developed from chemical analysis. However, bioaccumulation data alone do not provide evidence of biological consequences or effects upon the organism. The approach taken in this study was correlative in nature and did not seek to demonstrate cause and effect relationships from exposure to the complex of contaminants identified in BRH dredged material. The principal conclusions from these studies are summarized below.

- The principal residue-effect relationships were identified in the laboratory for scope for growth and histopathology measured in M. edulis; bioenergetics in N. incisa in the laboratory and in limited cases in the field. Residue effects relationships were also detected for adenylate energy charge in both M. edulis and N. incisa in the laboratory,

but the biological significance is not important since all the AEC values were within the normal range. Residue-effects relationships were detected for sister chromatid exchange in N. incisa in a single set of field samples.

- The most consistent correlations were achieved with PCBs and the higher molecular weight PAHs which are refractory to metabolic degradation and detoxification.

Recommendations

The following recommendations are presented in terms of the utility of the methods for predisposal evaluation of dredged material impacts and postdisposal compliance monitoring and assessment.

- Measures of mortality, scope for growth, reproduction, and population responses are recommended for inclusion in a hierarchal pre-disposal permit testing program. In addition to A. abdita, M. bahia, and M. edulis, consideration should be given to utilizing a suite of test species reflecting the local environment and representing a range of sensitivties.

- Measures of bioaccumulation are of fundamental importance to determining the bioavailability and exposure to contaminants in complex materials. Our experience suggests that test methods for both bedded and suspended sediment exposures are important. Both M. edulis and N. incisa performed well in these studies and should be considered in a suite of test species for bioaccumulation. The results indicate that test durations of 30 days or greater are necessary to properly quantify uptake of environmentally important contaminants.

- A genotoxic test method is also recommended for inclusion in the pre-disposal permit testing program. However, SCE is not adequately developed to fill this need. Although we did not do an in depth analysis of the Ames test, we suggest that it along with other genotoxic methods be evaluated for predisposal evaluation of sediments.

- The development, application, and evaluation of benthic assessment methods suggest that the REMOTS rapid reconnaissance technique be utilized during pre-disposal site evaluation. The results from this technique corresponded with conventional measures of benthic community structure and health.

- Several of the methods evaluated in this program have utility for field assessments. These include scope for growth, growth and bioaccumulation measured in M. edulis, and bioenergetics and bioaccumulation measured in N. incisa. Mortality and growth of A. abdita exposed to cored sediments from the disposal site were also utilized to determine the spatial and temporal changes in sediment toxicity. The

two benthic community assessment methods evaluated in this program are recommended for field assessments. The REMOTS method should be used as a rapid reconnaisance tool to identify and characterize the spatial and temporal extent of impacts and to direct more intensive sampling for detailed species ennumeration.

CONCLUSIONS

The conduct of this study has provided considerable insight into the application and field verification of biological test methods for the evaluation of dredged material. The conclusion reached is that there are several methods suitable for pre-disposal permit evaluation and post-disposal monitoring of dredged material impacts. The evidence presented shows that the biological responses observed in the laboratory can be expected to occur in the field when analagous exposure conditions are encountered. Residue-effects relationships were examined through correlation analysis. The most consistent correlations were achieved with PCBs and the higher molecular weight PAHs which were refractory to biodegradation. The analyses did not seek to demonstrate cause and effect relationships from exposure to the complex of contaminants identified in the dredged material.

The pro- and retrospective assessment of the fate of the dredged material and associated contaminants has lead to the following observations. Mass balance estimates of the dredging and disposal operations indicate that while 70% - 80% of the dredged material was accounted for, only 30% - 60% of selected contaminants reached the sea bed at the disposal site. The high organic content (6%), high water content (69%), and fine grained nature of this maintenance dredged material creates special problems for containment during both the dredging and disposal operations. Near-bottom contaminant transport occured 1000 meters from the disposal site but was detectable only during and for one month after disposal. Contaminant concentrations in the surficial sediments (0-2cm) at the disposal site decreased significantly after nine months due to natural geochemical and biologically mediated processes. However, contaminant concentrations in the deeper sediments remained at initial post-disposal levels for the duration of the study. Episodic events (hurricane Gloria) scoured the oxidized surficial sediments, re-exposing the deeper anaerobic contaminated sediment.

Biological effects, in the near-bottom water column, measured as scope for growth and bioaccumulation in M. edulis were detected during and for one month after disposal after which time no effects were detectable. Biological effects in the benthos, determined from bioenergetics, bioaccumulation, recolonization, and benthic community analysis persisted for up to one year on the perimeter (400 m) of the site where the depth of dredged material was 1-2cm and for three years at and within 200 meters of the site apex where the depth of the dredged material ranged from 10cm to >50cm. Of the specific biological responses measure in N. incisa only respiration and excretion rates in worms collected on the disposal site remained significantly affected for the three year study period. Tissue residues of PCBs and the higher molecular weight PAHs, in N. incisa, remained elevated from one to three years

after disposal depending on the distance from the disposal mound. Initially elevated PCB and PAH tissue residues in populations at 1000 meters from the apex and 600 meters from the perimeter of the disposal mound returned to background levels within one year. However, contaminant residues in worm populations collected at the perimeter of the disposal site (400m) remained elevated for two years and returned to background only after three years. N. incisa populations that repopulated a 200 meter radius of the mound apex a year after disposal had PCB residues (750 ng/g) or 3.5 times background which increased to 1000 ng/g after three years. This data suggests that N. incisa continued to provide a biological vector for deep sediment contaminants throughout the study.

Though individual biological responses from single species are useful indicators of potential environmental impacts, the ultimate ecological benchmark is the populations or communities at risk. The organism sediment index determined from REMOTS indicated that at and beyond the perimeter of the mound (>400m) the functional relationships between the sediment and biota returned to normal after 6-9 months. However, the mound (200 m radius from apex) did not return to pre-disposal geochemical nor biological conditions after three years. Specifically, the redox boundry did not penetrate beyond 2-4cm, and the Stage-III, deposit feeding molluscs (e.g. Nucula/Yoldia) had not become established three years after disposal. Lack of recovery on the mound is probably the result of the interactions among the depth, granulometry, geochemistry, redox state and contaminant concentration of the dredged sediment. The sum of such interactions affect the processes involved in recolonization and subsequent survival, growth and reproduction of the colonizing species. Individual species tests are unable to simulate these interactions and are thus unable to forecast community level impacts. However, the former are useful in identifying the potential for both direct toxic effects and for contaminant movement within the infaunal biota which leads through commercially important species to humans.

DISCLAIMER

This report has been reviewed by the Environmental Research Laboratory, U.S. Environmental Protection Agency (EPA), Narragansett, Rhode Island, and approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of the U.S. EPA, nor does mention of tradenames or commercial products constitute endorsement or recommendation for use. Support for this work was provided to K. John Scott and Wayne R. Munns through EPA contract #68-03-3236 to S.A.I.C.

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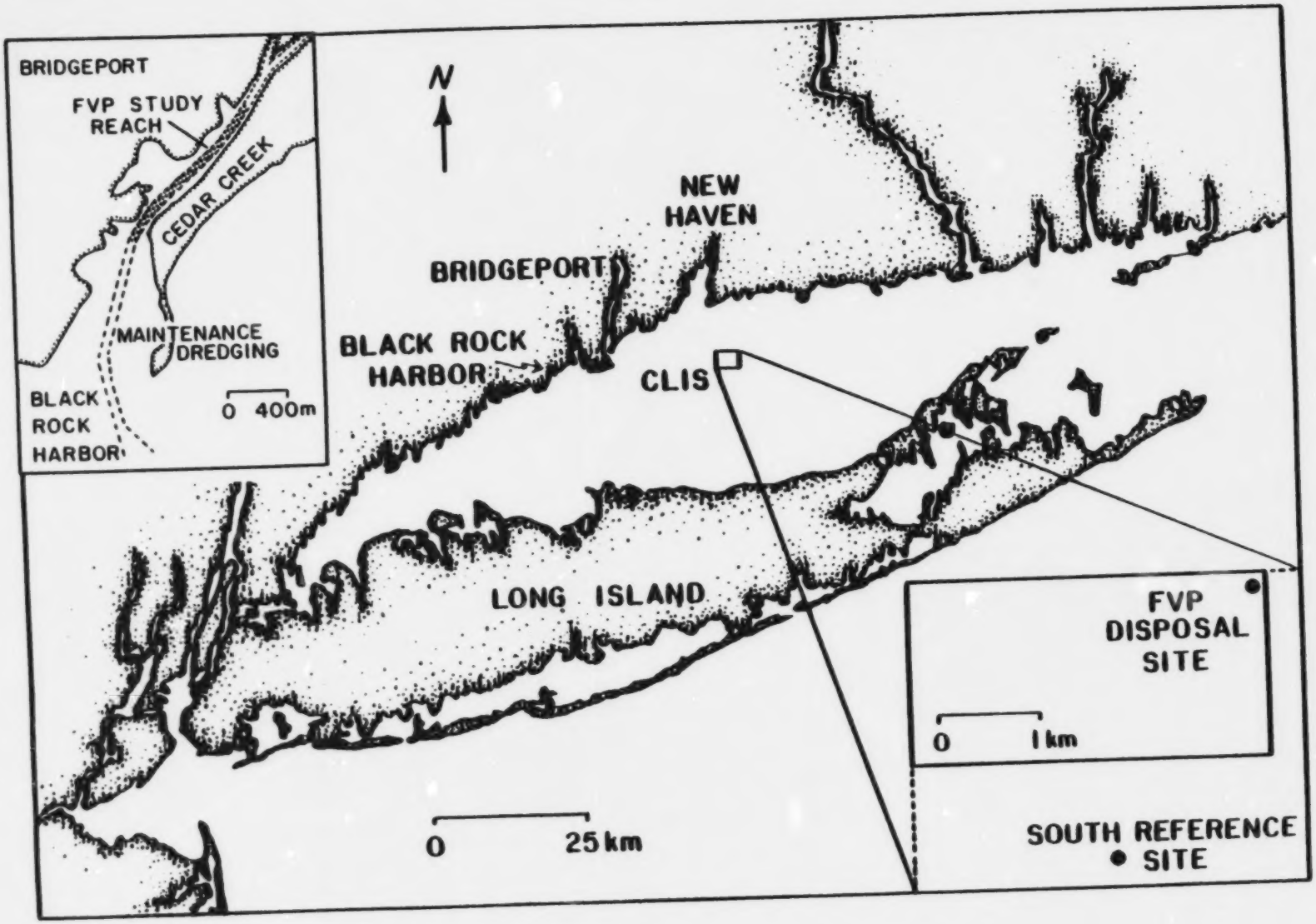


FIGURE 1. Central Long Island Sound disposal site on Black Rock Harbor dredge site.

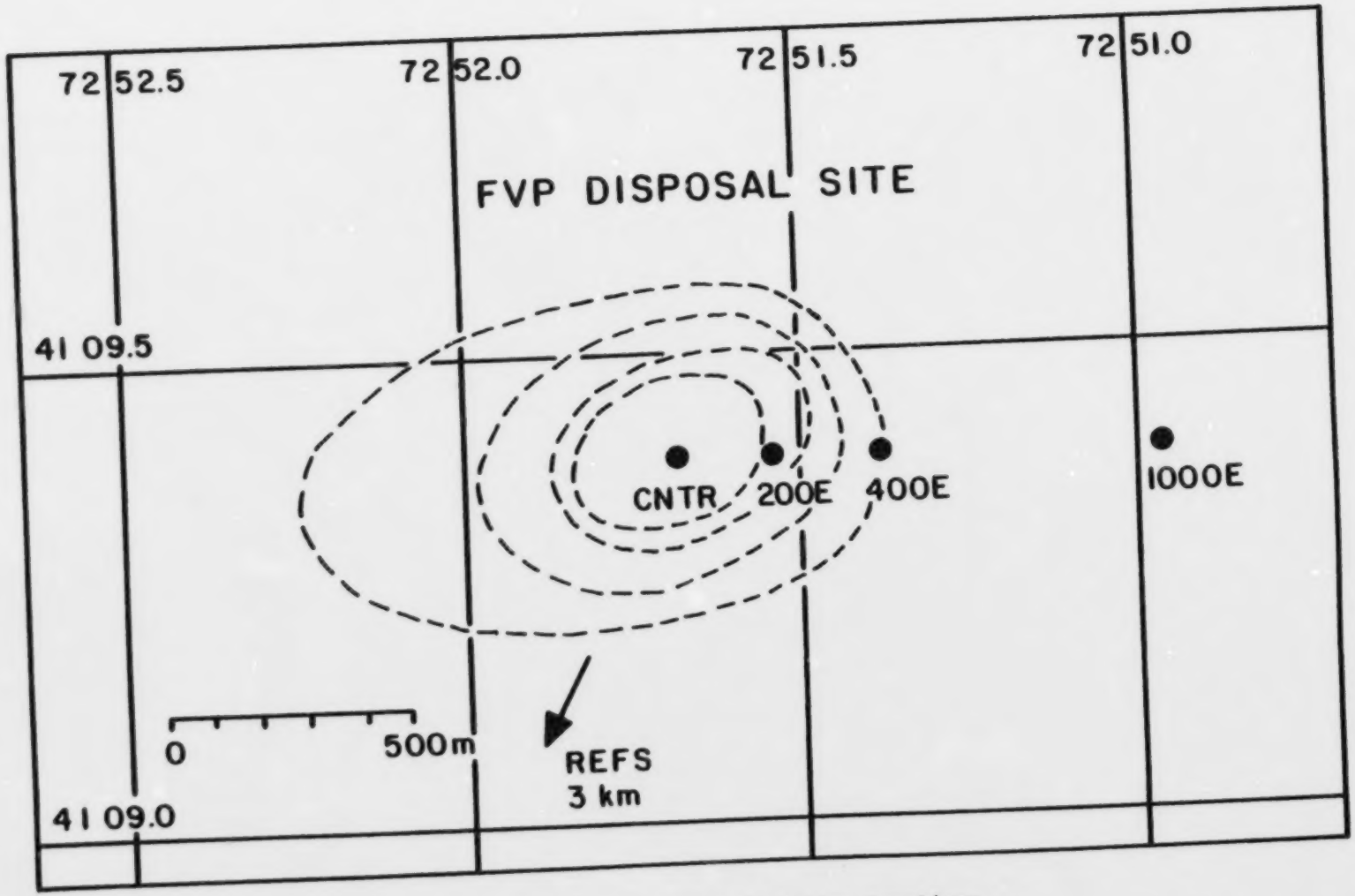


FIGURE 2. FVP disposal site and station locations.

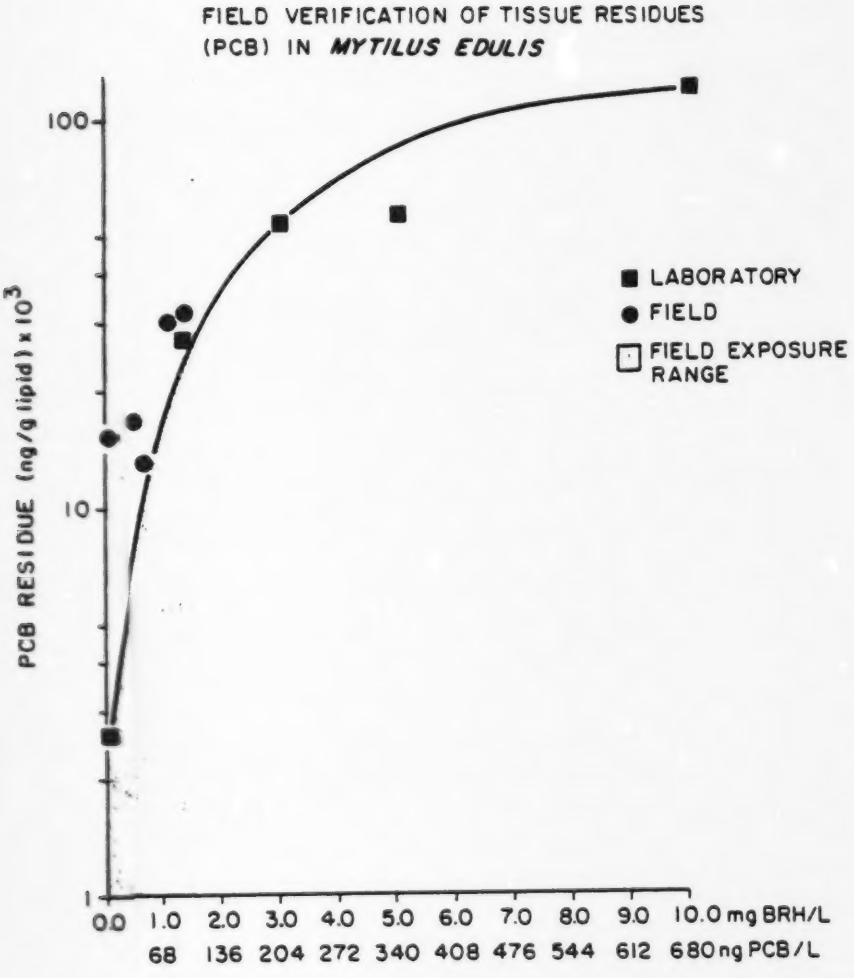


FIGURE 3. Field Verification of Tissue Residues (PCB) in Mytilus edulis

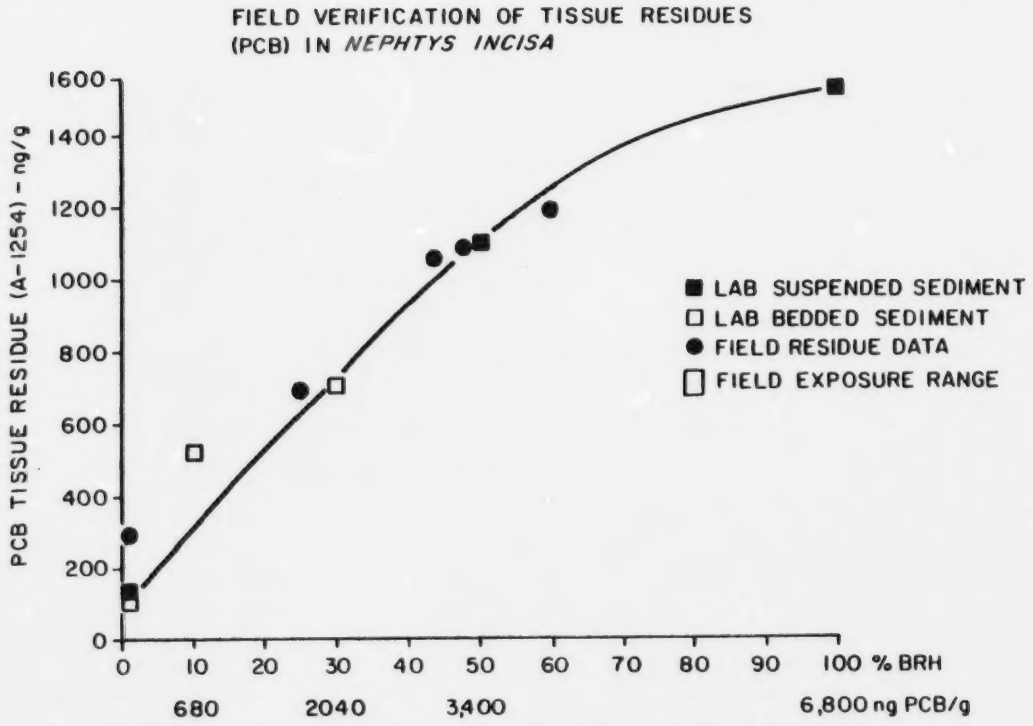


FIGURE 4. Field Verification of Tissue Residues (PCB) in Nephtys incisa

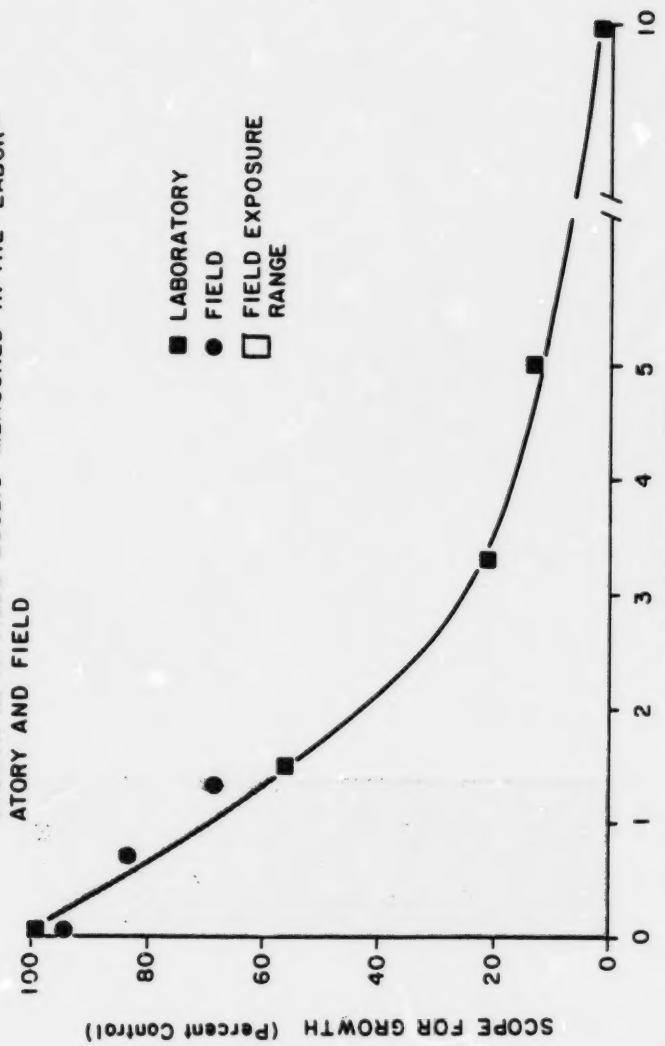


FIGURE 5. Exposure-Response Relationship for Scope for in Mytilus edulismeasured in the laboratory and field

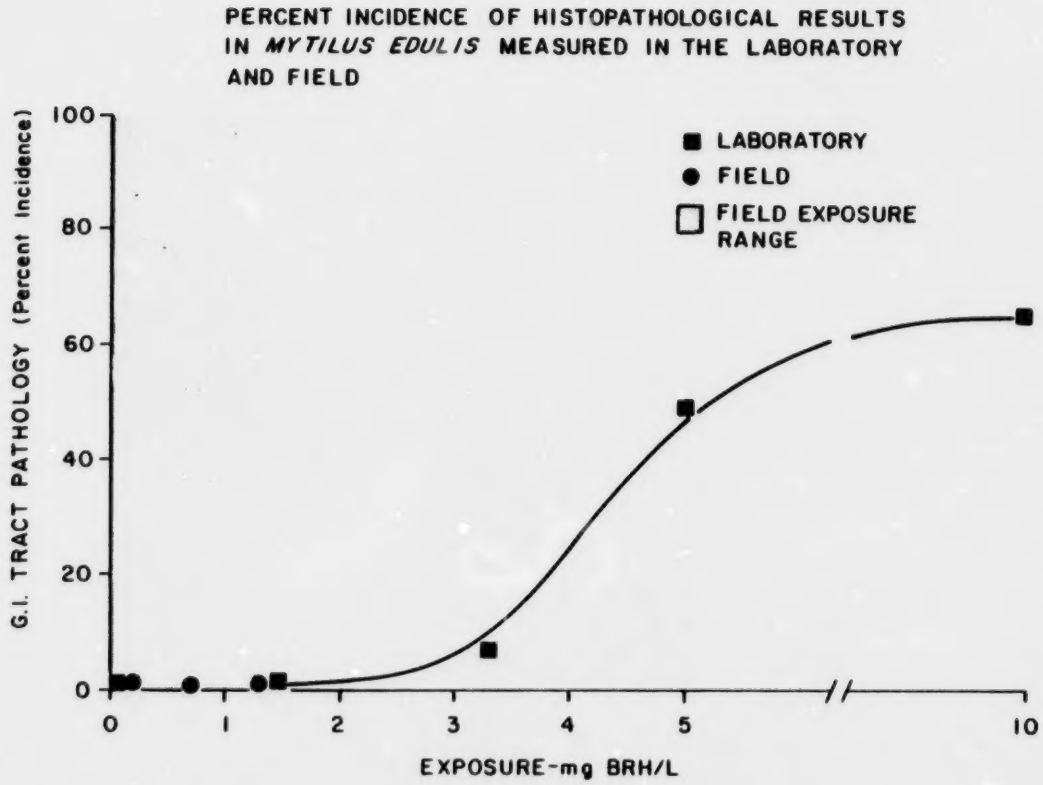


FIGURE 6. Percent Incidence of Gastiointestinal Tract Pathology in Mytilus edulis Measured in the Laboratory and Field

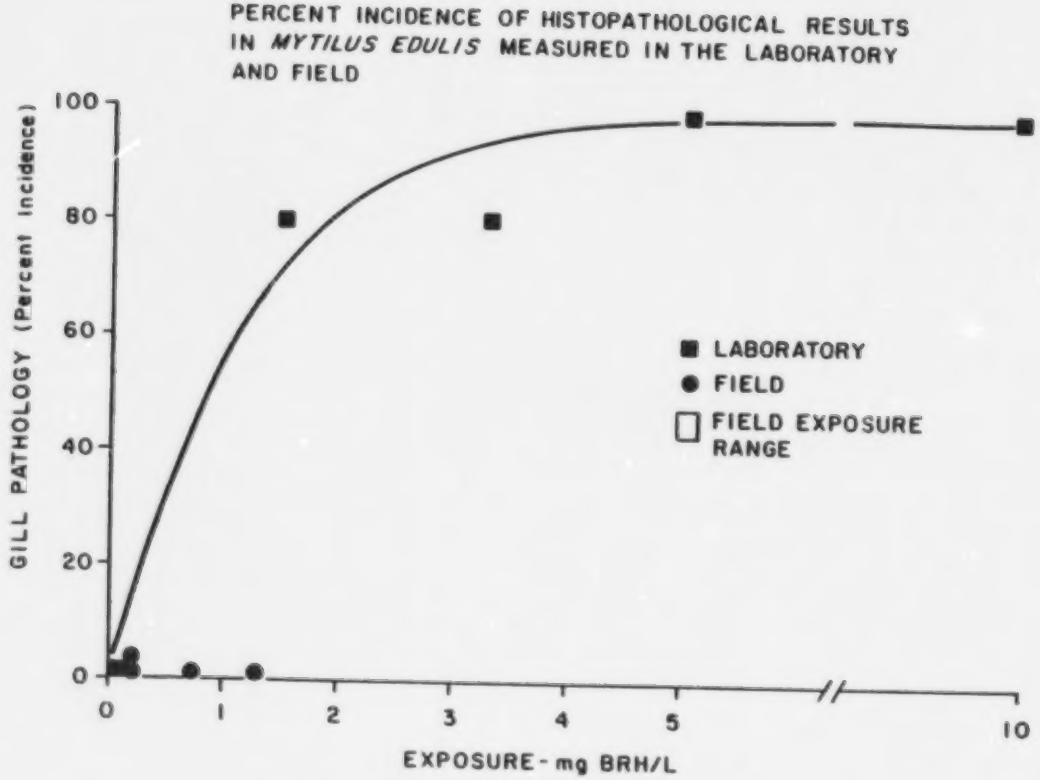
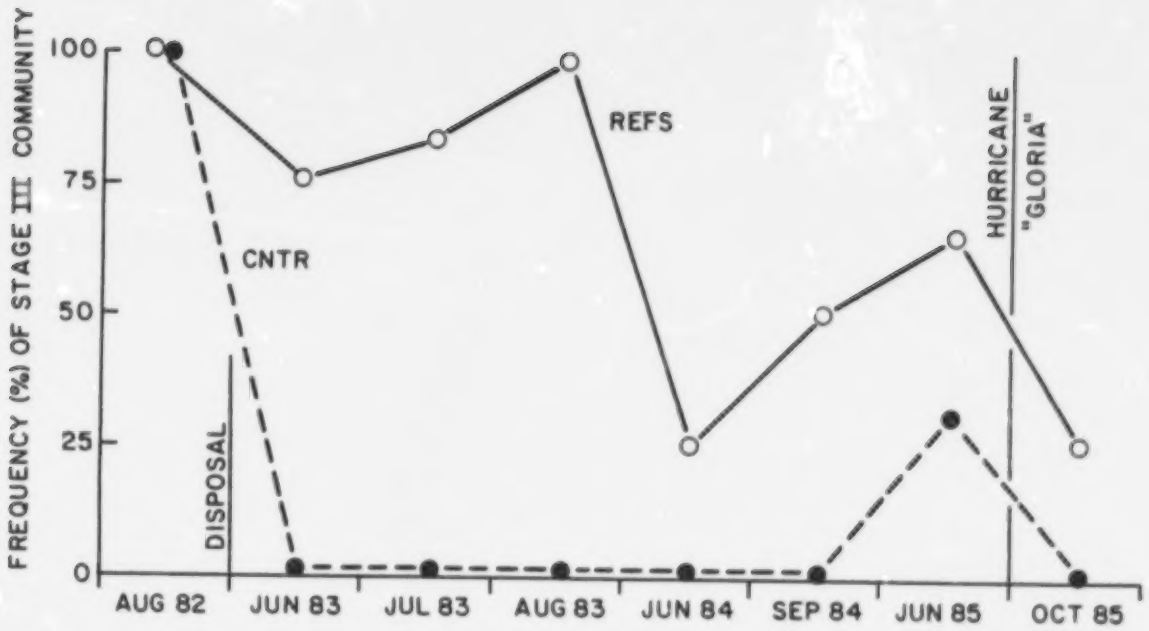


FIGURE 7. Percent Incidence of Gill Pathology in Mytilus edulis Measured in the Laboratory and Field



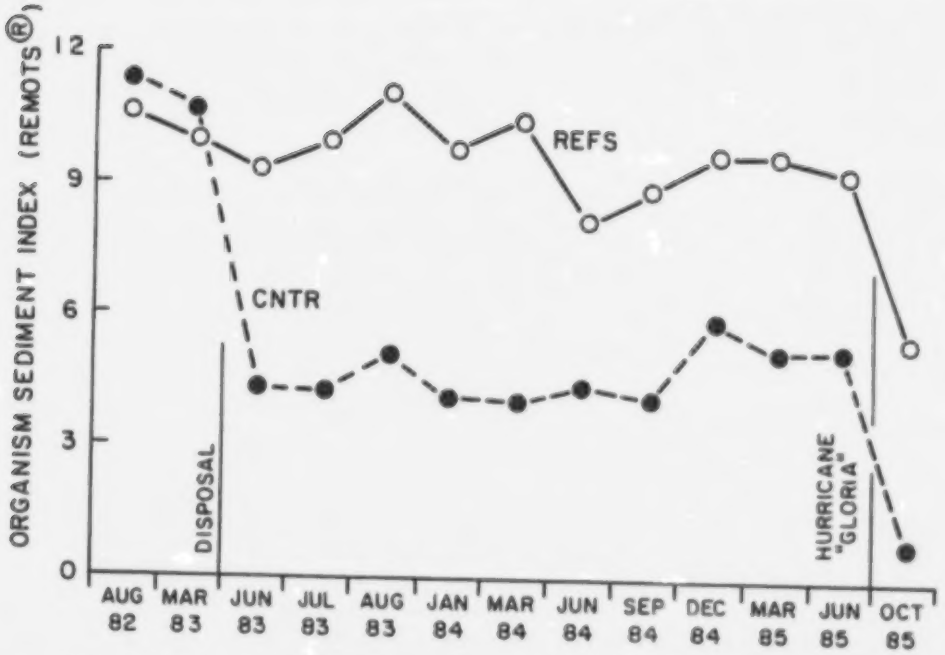


FIGURE 8. The Temporal and Spatial Pattern of Community Parameters, Successional Stage and Organism Sediment Index

FIELD VERIFICATION PROGRAM RESULTS OF WETLAND CREATION AND UPLAND DISPOSAL STUDIES

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Contaminated dredged material was tested in the laboratory under three disposal site environments; aquatic, intertidal, and upland. Laboratory test results were then verified in the field at aquatic, intertidal, and upland disposal sites. Results of the intertidal and upland disposal sites are discussed in this paper.

Laboratory tests with wetland plants and animals indicated freshly disposed dredged material was not toxic to plants but was toxic to soil-dwelling animals. Field verification of these results was obtained for one of two test plant species. Wetland animal test results were somewhat verified under field conditions. Toxicity of the disposed dredged material was observed only during the summer and not at the cooler seasons of the year. Wetland animals appeared to bioaccumulate certain metals, PAH's, and PCB's.

Laboratory tests with upland plants and animals were verified at the field site. Upland disposal of the contaminated dredged material resulted in death of both plants and soil-dwelling animals. Water quality test results in the laboratory were successfully verified under field conditions for upland disposal. Longer term predictions in

laboratory tests of contaminated dredged material placed in wetland or upland environments require consideration of the effects of ageing and leaching of excess salt from the disposal sites.

MANUSCRIPT NOT AVAILABLE AT TIME OF PUBLICATION.

FIELD VERIFICATION PROGRAM: SYNTHESIS AND COMPARATIVE ASSESSMENT OF DREDGED MATERIAL DISPOSAL ALTERNATIVES

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Methods presently used to evaluate the disposal of dredged material in the ocean and in alternative locations are based on first generation techniques which have not been adequately field verified. The U.S. Army Corps of Engineers (Corps) manages the disposal of dredged material based, in part upon criteria established in consultation and in conjunction with the U.S. Environmental Protection Agency (EPA). In order to uphold legal responsibilities, it is necessary to provide both the Corps and EPA with documented and field verified evaluative procedures upon which environmentally sound decisions regarding the disposal and management of dredged material can be made.

The Field Verification Program (FVP) was designed to help meet this critical need. The purpose of the FVP is to document and field verify existing methods used to evaluate the effects of contaminated dredged material disposal in oceanic, nearshore and upland environments. The Corps' Waterways Experiment Station, lead laboratory for the FVP, conducted the upland and nearshore disposal site studies. EPA's Environmental Research Laboratory at Narragansett, RI carried out work at the aquatic disposal site. Detailed descriptions of their respective investigations can be found elsewhere in these proceedings (see C.R. Lee *et al.* and J.H. Gentile *et al.*).

The FVP was authorized and funded in 1982. Sediments from Black Rock Harbor (BRH), CT, were characterized physically, chemically and biologically prior to dredging operations in the spring of 1983. The majority of BRH sediment was disposed of in the ocean. A portion was used to create a wetland environment and the remainder was placed in an upland disposal area. To our knowledge, this is the only instance in which contaminated sediment from a single waterway was placed in the three disposal environments and evaluated concurrently.

A comparative assessment of the effects of placing BRH sediment in these three disposal environments will be presented. This evaluation will be based on four fundamental observations; 1) physico-chemical changes in BRH sediment after disposal, 2) potential contaminant mobility, 3) exposure-response relationships in biota and 4) the spatial and temporal distribution of contaminants.

MANUSCRIPT NOT AVAILABLE AT TIME OF PUBLICATION.

EVALUATION OF THE FISH COMMUNITY IN SELECTION OF DISPOSAL SITES FOR DREDGED MATERIALS IN THE LOWER BAY COMPLEX OF THE HUDSON-RARITAN ESTUARY

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A series of fishery surveys was made by trawling through the Lower Bay Complex of the Hudson-Raritan estuary system, as part of the site selection process to implement a program for disposal of large volumes of dredged material in Lower New York Harbor. The fish catch data were from 15 operated surveys of 38 trawling stations made between September 1984 and March 1985. Catch data were standardized as fish densities (species abundances per hectare) and processed for the evaluation. The information examined included species composition, dominance, abundance, species richness, diversity and equitability. This information was compared station by station, and the comparisons used to distinguish within the Lower Bay Complex areas of relatively low diversity and low use by the fish community; four such areas were identified. Seasonal catches in these low use areas were further inspected for selected resource species considered of importance to the commercial and recreational fisheries.

Burial of dredged sediments in deep subaqueous borrow pits in the Lower Bay Complex is a possible disposal option. Using the information from the catches during the trawl series survey, the fish populations sampled in three existing borrow pits were compared with the fish populations in neighboring areas surveyed at the same time. The borrow pits sampled

were found to contain relatively diverse and dense populations of fishes, they were used much more by the fish community than were shallower areas sampled in the vicinity.

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REDUCTION AND RESOLUTION OF CONFLICTS BETWEEN OCEAN DUMPING AND AQUACULTURE

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ABSTRACT

The recognition of aquaculture as a legitimate use of the nearshore marine environment and the dramatic growth of aquaculture activity in Atlantic Canada over the past ten years has resulted in an increase in conflicts between aquaculture operations and ocean dumping. In addition, whereas ocean dumping in the Maritime provinces is administered from a single office, aquaculture leasing and licensing is conducted from four different offices in three provinces. This paper discusses interaction between various components of both of these activities.

The initial indications of potential conflicts between ocean dumping and aquaculture came as a result of an allegation that a dredge spoil plume had impacted a mussel farming operation. Since that time, the resuspension of contaminants and faecal coliforms have been raised as issues. In many cases, the dredging activities, which are the principle source of conflict, are being conducted on behalf of Small Craft Harbours, Department of Fisheries and Oceans, as a result of requests by local fishermen and aquaculturists. The continuous, year-round nature of some of these aquaculture operations precludes the use of simple changes in dredging schedules as a mitigative measure. The potential for elevated levels of conflict is increased by the fact that the very features which make public harbours suitable for their purpose, also makes them attractive to aquaculturists.

A computer-based system used to identify potential conflicts during ihe permit application process has been developed. The system allows an operator possessing the geographic coordinates of a proposed project to search a complementary database for activities falling within a variable-sized quadrat which could conflict with the proposal.

While incorporating dumpsite data from all of Atlantic Canada, the database presently manages only approximately 200 aquaculture leases and licenses in Nova Scotia. Expansion to cover an additional 3500 aquaculture operations in the other Maritime Provinces may take place in 1988.

Introduction

When activities sharing common requirements are undertaken in close proximity to one another, the potential for conflict arises. Ocean dumping, particularly of dredge spoils, and aquaculture are two such activities. In this paper, I wish to explore the attributes of these two activities which predispose them to interaction; examine the nature of these interactions/conflicts; explore possible solutions; and explain how the Marine Environment Branch of Environmental Protection in Atlantic Canada has begun to develop a strategy for resolution of these conflicts.

Why is aquaculture so prone to conflict with ocean dumping?

The selection of permanent sites for sustained human activity is generally based upon criteria which are used to evaluate the various facets of a site's suitability for that activity. These site selection criteria may be quite informal and applied in an *ad hoc* manner as was quite likely the case during the early settlement of fishing communities in Atlantic Canada, or may be part of a formal process employed by planners. Regardless of formality of the application, if we compare the priority criteria used in siting fishing harbours and aquaculture ventures, we can begin to see how competition for access may arise.

Criteria	Fishing Hr.	Aquaculture
shelter	very sheltered	very sheltered
water depth	navigable	navigable
market access	necessary	necessary
near grounds	necessary	n/a
clean water	for processing	necessary
industrial	low	low

It is quite understandable, considering the similarity of the activities, that some geographical overlap is possible. If we add to this arguement the fact that aquaculture operations actually benefit from the proximity to fishing harbours through improved access to fisheries-related infrastructure such as wharves, ice-making facilities, roads, electric power and processing plants, interactions between the industries become inevitable.

For various reasons, a significant proportion of fishing ports in Atlantic Canada require maintenance dredging in order to preserve navigability. It is this link which brings ocean dumping into interaction with aquaculture.

What is the nature of the conflicts between ocean dumping and aquaculture?

Dredging and dumping of dredge spoils have a potential

to impact aquaculture operations in three principle ways:

1. Through various mechanisms, cultured organisms can suffer mortality;

2. Several features of some dredged material can influence palatability, market acceptability, or safety of cultured organisms;

3. Regardless of actual impacts on a culture operation, expectations or perceptions of an impact by an operator or the public can result in conflict with a dredge operation.

Dredge-related mortality: The likelihood of dredging or dumping operations directly resulting in mortality of cultured species is very low. The most probable agents nonetheless are sediment load in the water column and oxygen starvation due to oxygen scavenging which can occur during oxidation of highly organic re-suspended material.

Elevated suspended particulate matter can cause mortalities in three ways. First, when the material settles out it can physically cover and smother benthic organisms. The three species most likely to be affected are oysters, mussels, and scallops. Of these, only oysters are presently cultivated extensively on-bottom. Recent research indicates that bottom culture may also be a viable option for mussels.

Second, silt has been shown to obstruct and/or damage gill lamellae in teleosts. This is a particular risk to young fish and to fish such as trout and salmon held in cages. Studies have shown that the nature of the suspended material is particularly important. Sharp particles cause lethal abrasion at low concentrations, while fine, soft particles can be tolerated at much more elevated levels.

Finally, elevated levels of suspended particulate matter have been demonstrated to cause bivalve molluscs to close their shells for long periods. If this occurs at a time when the animals are already stressed, such as immediately after summer spawning at a time when water temperatures have peaked and the carrying capacity of water for oxygen is reduced, mortality can result. This phenomenon, called "summer kill" is not unusual in some areas even in the absence of dredging. Of these mortality-inducing factors, the last is likely the most serious.

Dredging-related quality degradation: Dredging and ocean dumping activities can cause degradation of the quality of aquaculture stock in three principle ways: dredging can introduce material such as PCB's and heavy metals into stock which makes them unsafe to eat; suspended dredge spoils can reduce the market acceptability of aquaculture products by introducing grit, off flavours, or decreasing shelf life (particularly of species sold live such as mussels and

oysters); and dredging can resuspend bacteria which show up as coliforms during fish quality inspections, rendering the product unmarketable.

Perceptions: The most difficult issue to resolve of all of these is that of perceived impacts or expectations of impacts. We, as dumping regulators or operators, deal on a daily basis with reports of PCB's, heavy metals, DDT and a host of other scheduled or restricted substances. In the vast majority of cases in eastern Canada, levels of contaminants are found at background levels which pose no immediate threat to public health or marine biota. Nevertheless, our admission of their presence, even in minute amounts, triggers an entirely different response in most other people. In addition, our inability to quote exact LC50's of suspended particulate matter for their species of interest, despite the host of other uncontrollable environmental factors encountered on aquaculture sites, does nothing to help aquaculturists accept our reassurances of minimum risk regardless of how safe the operation may be.

Significant alarm has been raised in several instances over alleged or expected impacts on aquaculture operations. While direct liability is almost impossible to prove, it is equally impossible but equally important, particularly in the case of government operators, to prove their innocence if for no other than political reasons.

What are some possible solutions to these conflicts?

The resolution of conflicts between ocean dumping and users of common property resources is usually relatively easy. In most cases, the target species is transient and therefore only fished at a particular location for part of the year. In order to avoid conflict, it is necessary only to schedule dredging during the off-season.

On the other hand aquaculture stock, being owned property and being resident in cages or on fixed substrate, is much less amenable to such simple solutions. A typical but by no means exhaustive list of solutions in a regulator's repertoir might include the following:

1. Move them - In some cases where the distances are not great, the dredging schedule is not protracted, and the aquaculture facility not elaborate, it may be cost effective to pay an aquaculturist to relocate his stock to a more safe location. There are risks to both stock and equipment in this option.

2. Timing - Timing can be used in two ways to reduce risks to aquaculture operations. Dredging or dumping can be limited to tide states which carry resuspended material away from the site. Alternatively, some aquaculture production schedules

such as sea-cage rearing of trout lend themselves to seasonal rescheduling of dredging/dumping.

3. Silt curtains, deflectors, etc. - Physical structures, most often associated with the dredge/dump operation, can prove effective in controlling extensive migration of suspended material.

4. Change methodology -A simple change of equipment or operating procedures employed by the dredging operator can significantly reduce risks.

5. Avoidance -Two options exist under this category. First and probably most frequently used is for the dumping operation to be relocated to a more appropriate site. The second, and rather more difficult option to employ, is to designate dump sites of very high importance and heavy use and prohibit aquaculturists from using the area.

6. Compensation -If all else fails, the most cost effective solution to conflict may be compensation for damages. Be forewarned, aquaculture stock is often worth more in the water than on the table.

How Environmental Protection - Atlantic Region is developing a strategy for resolution of conflicts between ocean dumping and aquaculture.

Although the potential for conflicts between ocean dumping and aquaculture had been recognized long before, the seriousness of not resolving them in advance only became obvious when a regulating agency in the Maritimes was accused of not upholding its responsibility to protect an aquaculturist after he suffered losses allegedly related to an ocean dumping operation. As a result, Marine Environment Branch (MEB) of Environmental Protection was given the responsibility of developing a strategy for dealing with aquaculture on a regional basis.

MEB is now the single window through which Conservation and Protection, Environment Canada responds to the aquaculture industry on a regulatory basis. MEB presently plays a formal role in evaluation of all aquaculture licence and lease applications in Nova Scotia. It is also presently in discussions with New Brunswick and Prince Edward Island to establish a contributory role in their regulatory processes. In return, MEB will receive access to information on prospective aquaculture operations so that ocean dumping conflicts can be avoided.

Being involved on both sides of the regulatory process - aquaculture site leasing and ocean dumping site evaluation - puts the Branch in a unique position to deal with this conflicts issue.

The first step in conflict resolution is identifying where conflicts are likely to arise. For this purpose, two databases have been developed using the Branch's IBM PC and the database manager "Paradox". One database holds geographic, technical and chemical data for all ocean dumping projects in Atlantic Canada for the past three years. The other database, which is nearing operational status, holds geographic and technical data on all aquaculture permit, licence, and lease applications in the province of Nova Scotia.

Through Paradox, reciprocal queries between these two databases can be conducted during evaluation of applications for either activity. Generally, queries are conducted using variable-sized search quadrats based on an operation's latitude and longitude. Potential conflicts, identified as the presence of another potentially incompatible activity within the search quadrat, are brought to the attention of MEB staff. The information is then transmitted to both the ocean dumping operator if it is an active ocean dumping permit file and the aquaculturist. Once a potential conflict arises, it is usually left to the two parties involved to negotiate a resolution, but in particularly difficult cases MEB has and will continue to take an active role in this process.

MEB's strategy continues to evolve as conditions change. We can only hope that as the potential for conflict increases, our capability to foresee and resolve these issues will expand and mature.

THE TAMPA HARBOR PROJECT: A MAJOR MONITORING SUCCESS

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Abstract

The Tampa Harbor Project was a major dredging project for Tampa, Florida, that widened and deepened the existing shipping channel. The Project began in 1975, and was completed in 1985. Approximately 70 million cubic yards of dredged material were removed from the existing channel; the majority of the material was placed adjacent to the channel. However, approximately 8.4 million cubic yards were ocean-disposed. The Environmental Protection Agency and the Corps of Engineers were involved in all aspects of the Project, and a number of local and State organizations were considerably involved in portions of the planning process.

This paper will review all aspects of the Tampa Harbor Project, from the initial disposal operations at Site A, the litigation initiated by Manatee County, the subsequent trial and judicial outcome, the numerous surveys for an alternate disposal site, the formulation of the monitoring and management plans, the selection of a new disposal area at Site 4, and the resumption of disposal operations. It will also discuss in detail the scientific data, results, and conclusions of the monitoring operations, including the emergent biological growth on both Sites A and 4, following cessation of disposal operations.

I. Introduction

The Tampa Harbor Project involved deepening and widening approximately 50 miles of existing shipping channel in Tampa Bay, Florida, with the removal of approximately 53 million cubic meters (70 million cubic yards) of dredged material (Figure 1). The channel runs from the Port of Tampa south and west through Tampa Bay, and extends approximately 13 nautical miles (nmi) beyond Egmont Key at the entrance to

the Bay. The deepening project allowed deeper draft freighters to utilize Tampa Bay; in the past, these vessels were often required to leave Tampa Bay ports lightloaded, due to insufficient channel depths for fully-laden ships. This project by the Corps of Engineers was begun in 1975, and was completed in November 1985. Some of the removed dredged material was disposed of adjacent to the main channel, and some was used to construct diked disposal lagoons in upper Tampa Bay; however, a substantial amount of the dredged material was barged beyond the limits of Tampa Bay for ocean disposal. The material disposed of is geologically laid-down consolidated clays, silts, and sands present since the formation of Tampa Bay approximately 20,000 years ago, due to the rise in sea level following the last Ice Age.

II. Background

The Environmental Impact Statement (EIS) for the Tampa Harbor Project (required under the National Environmental Policy Act of 1970) was prepared by the Corps of Engineers in late 1974; although dredging operations began in 1975 with disposal of dredged material within Tampa Bay, it was not until June 1980 that the Corps began disposing of dredged material at an ocean dredged material disposal site (ODMDS). The ODMDS that the Corps initially utilized was a square site 0.82 nmi on a side known as Site A, which is approximately 13

nmi west of Egmont Key. Shortly after ocean disposal began, several parties in Manatee County (the political subdivision covering the southern half of Tampa Bay) alleged that the disposal operations were damaging productive sponge and hard and soft coral bottoms, which provided habitat for fish and invertebrates, and that these habitats, beyond the limits of the disposal site, were being buried by the disposal operations.

In April 1982, Manatee County and several other plaintiffs initiated litigation against the Environmental Protection Agency (EPA) and the Corps to bring the disposal of dredged material to a halt. The plaintiffs alleged that the disposal operations were adversely affecting valuable fishing and SCUBA diving activities, and that the material being disposed of was spreading beyond the limits of Site A, causing continuing and irreparable damage to the productive habitats beyond the boundary of the site. In November 1982, the litigation initiated by the plaintiffs was heard in Federal District Court in Tampa; on December 21, 1982, the court halted any further disposal of dredged material at Site A by the Corps after December 24, 1982. The court stated that two of the reasons for its decision were that the environmental impact of dredged material disposal at Site A had not been adequately considered, and that a site-specific survey of Site A prior to initiating disposal of dredged material there had not been performed. Site A received 3.78 million cubic meters (4.94 million cubic yards) of dredged material between June

1980 and December 24, 1982; no disposal has occurred at Site A since the termination of disposal operations there.

Beginning in October 1981, and continuing in May 1982, February/March/April 1983, and September/October 1983, EPA performed surveys of potential alternative ODMDS's in the Gulf of Mexico west of Tampa Bay. During the May 1982 survey, an extensive sandy-bottomed area approximately 18 nmi west-southwest of Egmont Key was discovered. This area, designated Site 4, was established as a square site, two nmi on a side, and was found to be virtually barren of hard bottom areas or sponge and coralline growths. During the February/March/April 1983 survey, Site 4 was examined in extensive detail with an underwater videotape camera; over 35 nmi of videotape data within and immediately surrounding Site 4 was collected.

The videocamera transects revealed that the vast majority of Site 4 had a minimum of hard bottom areas, and was characterized by flat, barren, sandy areas occasionally interspersed with 2.54- to 15.24-cm (1- to 6-in) high sand waves interdigitated with shell hash. Analysis of the videotape of all of Site 4 demonstrated that over 83 percent of the area viewed was virtually devoid of any form of coralline or sponge communities, approximately 17 percent of the site had sparsely covered hard bottom areas, and less than one percent

of the site could be characterized as heavily populated hard bottoms.

Further studies and analysis of the site revealed that Site 4 is located in a water depth of 20 to 25 m (66 to 82 ft) on the west Florida continental shelf, a broad carbonate plateau overlain by a veneer of predominantly carbonate sand and shell. Where the sand layer is sometimes thin near shore, hard bottom is occasionally exposed in the form of low-relief outcrops, ledges, or patch reefs, some of these being remnants of ancient shoreline features. More extensive areas of irregular hard bottom are covered by a mobile sand layer, which varies in thickness. Sessile epifauna such as corals and sponges are associated with both emergent and sand-covered hard bottom. Benthic macroalgae are also widely distributed in both hard-bottom and soft-bottom areas on the inner shelf.

In October 1982, EPA issued a draft EIS on Site 4, and in November 1982 proposed the designation of the site for the disposal of dredged material. Following the Agency's assessment of the comments received during the comment period, EPA issued a final EIS on Site 4 in September 1983, and in November 1983 designated Site 4 for the disposal of dredged material from the Tampa Harbor Project for a period of three years (Figure 2).

The designation notice (48 FR 50317, 1 November 1983) stated that the Agency intended to "carefully monitor the effects of disposal operations at Site 4 to assure that no significant adverse environmental effects occur beyond the boundaries of the site." The notice further stated that a monitoring plan, including the necessary parameters and their limits, would be developed by representatives of the EPA, the Corps, the Tampa Port Authority, and State and local Florida officials and scientists.

That monitoring plan was finalized in February 1984; the primary emphasis of the monitoring plan was to determine the movement of the disposed dredged material within Site 4, although studies on the environmental impact of the dredged material were also a major part of the plan. The plan called for quarterly surveys for a period of three years; the first monitoring survey to Site 4 occurred in April 1984, with the emplacement of monitoring equipment.

III. Methodology

Disposal operations at Site 4 began in late May 1984; a total of 2.63 million cubic meters (3.44 million cubic yards) of dredged material were disposed of at Site 4 by the time the project was completed in late November 1985. The dredged material built a mound that was 1524 m (5000 ft) long on an east-west axis, 244 m (800 ft) wide on a north-south axis,

and approximately 12.2 m (40 ft) high. A cross-section of the mound is shown in Figure 3. The monitoring methodology included the use of station marker tripods, sediment traps, and current meters, as well as photographic quadrats. In addition, studies included analyses for two tracer substances (strontium and phosphate), grain size, and total organic carbon to differentiate between the dredged material and natural background sediments. The emplacement of all monitoring equipment and sample collections was performed by divers using SCUBA equipment.

During the initial April 1984 monitoring survey, the following equipment was deployed at 14 stations at Site 4 and at three stations at a reference site, which was five nmi to the southeast, in the same depths of water.

A) Station marker tripods

These were constructed from 2.54 cm (1 in) diameter construction reinforcing rods, with a vertical member 2 m (6 ft) in length painted in alternating bands of red and white every 12 inches, and with three members welded to the central shaft at angles of 120 degrees to form a stable tripod (Figure 4).

B) Sediment traps

These were constructed from one-liter screw-top polypropylene containers, fitted with a 2.54 cm (1 in) inner

diameter by 24.1 cm (9.5 in) tube at the mouth of the container. The aspect ratio of the tube was thus close to the optimal 1:10 ratio; this is important so that currents do not carry out sediments previously collected in the trap. The top edges of the mouth of the tube were bevelled away from the inner diameter, to avoid having sediments settling on the top lip of the tube from being potentially carried into the trap. The mouth of each trap was about 1 m (3 ft) above the seafloor. Two traps were attached to each of the station marker tripods.

C) Pingers

Helle Model 2260 transmitting pingers with Helle Model 1116 battery packs were attached to the station marker tripods to enable divers to more easily locate the tripods on subsequent monitoring surveys. The pingers transmitted on a band of 37 kH, once every second, and had a transmitting life in excess of six months. One pinger was attached to each of the tripods. Antifouling paint was applied to all station tripods and sediment traps to reduce algal growths on the equipment.

D) Current meters

One Endeco Model 105 current meter with recording film pack was deployed at each of two stations at Site 4, and one meter was deployed at the center of the reference site. Each

meter was attached to a tautline array at a height of 1 m (3 ft) above the seafloor, and set to record at one hour intervals. Because the integration time on the meters was doubled from the normal 30 minutes to one hour, the meters provided a recording duration of approximately 90 days.

In addition to the physical equipment deployed at Site 4 and the reference site, the following monitoring methodologies were also employed.

E) Photographic quadrats

A Nikonos Mk. III 35-mm underwater camera, equipped with a 28-mm lens and two 150 watt-second strobes, suspended in a steel frame 81.3 cm (32 in) above the bottom, was used to take quadrat photographs at two stations within Site 4, and one station at the reference site. Photographs of the seafloor and epibiota were taken along five 5 m (16.4 ft) transects radiating from the vertical shaft of the station marker tripods. Photographs were taken along the entire length of each transect; i.e., successive photographs along each transect were contiguous. A marker board showing the month, year, station, transect, and photograph number was placed within the field of view of the camera to identify each photograph.

F) Surficial sediments

Three replicate sediment samples were collected at each Site 4 and reference site station on each survey. Samples

were also collected on three surveys at two points in the disposal area, and on one occasion, from a disposal barge in Tampa Bay. The samples from the disposal area and the barge were obtained to provide information on the composition of sediments known to contain dredged material.

Sediment samples were obtained by sliding a plastic container along the seafloor surface and scooping out the top 2 cm (0.79 in) layer until a volume containing approximately 1 kg (2.2 lb) of sediment had been collected. The sediment samples collected were used to perform trace metals and geochemical analyses for strontium, phosphates, grain size, and total organic carbon.

G) Infaunal sampling

Ten infaunal samples were collected at each station during each survey. The corers used to collect the samples were 12.5 cm x 12.5 cm (4.9 in x 4.9 in) in surface area and 23 cm (9 in) in depth. A 0.5-mm mesh stainless steel screen covered the top of each sampler to prevent loss of macroinfauna. Divers collected samples within 5 to 15 m (16 to 49 ft) of the station marker tripods, depending upon location of sediment thick enough to allow adequate corer penetration. Divers pushed each corer into the sediment to a depth of 15 cm (5.9 in), then removed the corer by excavating the sediment away from one side, pushing one hand beneath the penetrating

end to cover it, and lifting the corer out. Each corer with its sample was placed in a fine-meshed cotton bag, which was then tied closed. Two corers were placed in each of five nylon bags, which were clipped together and carried to the surface by the divers.

IV. Results

The data from the current meters can be summarized as follows. Currents flowed predominantly toward the east during spring and early summer of 1984, then shifted toward the southeast and south during fall, winter, and spring of 1985. The strongest currents (0.45 cm/sec or 0.87 knot) occurred during late fall and winter, and were directed south. Most current velocities were less than 10 cm/sec (0.19 knot), and only rarely did they exceed 20 cm/sec (0.39 knot). The mean current velocity for the entire measurement period was between 5 and 8 cm/sec. Progressive vector calculations indicate that over the period of monitoring of Site 4, suspended particles would be transported to the east, southeast, or south (90° to 180°); however, the current velocities measured at Site 4 have little potential for eroding or resuspending sediments present on the seafloor.

Analyses of phosphate and strontium in the fine fraction of surficial sediments from Site 4 and in the fine material contained in the sediment traps showed that these tracers were very useful in

indicating the presence of dredged material. The concentration of phosphate in the dredged material ranged from 11,900 to 22,200 ppm; in contrast, the concentration of phosphate from Site 4 in the pre-disposal survey ranged from 3000 to 3200 ppm. Thus, high readings of phosphate in surficial sediment or sediment trap samples indicated the presence of dredged material.

On the other hand, the concentration of strontium in the dredged material ranged from 140 to 550 ppm; in contrast, the concentration of strontium from Site 4 in the pre-disposal survey ranged from 2470 to 2760 ppm. Thus, low readings of strontium in surficial sediment or sediment trap samples also indicated the presence of dredged material.

Although dredged material (as indicated by high concentrations of phosphate or low concentrations of strontium) was found to be present at several of the monitoring stations on a number of the surveys, no effect of the dredged material, as indicated by either phosphate or strontium, could be found on the hard bottom assemblages at Site 4.

The total mean infaunal abundance at Site 4 stations ranged from 2,176 to 17,472 individuals/m^2. Annelids contributed about 60% of total individuals during all surveys; arthropods were the next most abundant group (generally 15 to 30% of the total). Total infaunal biomass at Site 4 stations ranged from 1.06 to 16.94 grams wet wt/m^2. Annelids

were consistently the major biomass contributors; arthropods and molluscs were secondary contributors. The infaunal sampling and analysis did not reveal any effects of dredged material disposal at the Site 4 monitoring stations on the normally-present infaunal populations. Although significant differences in abundance and biomass between some Site 4 stations and reference site stations were seen on several surveys, it was impossible to attribute those significant differences specifically to dredged material. In addition, most significant differences in abundance and biomass were not accompanied by the appropriate high sediment concentrations of phosphate or low sediment concentrations of strontium, which would have indicated the presence of dredged material.

The use of photographic quadrats at Site 4 also was demonstrated to be a useful monitoring methodology. Although the hard bottom (sponge and coralline) growths at the stations where the quadrat photographs were taken were generally a low percentage of total biotic cover, there was no evidence that the fine material winnowed from the dredged material diposal operations caused any permanent damage to those hard bottom assemblages. Sessile epifauna, such as those corals and sponges found at Site 4, generally tolerate a certain amount of sedimentation as part of their natural environment; however, there is no evidence that the dredged material disposal operations at Site 4 permanently affected any of the sessile epifauna there.

V. Conclusions and Recovery of Site A and Site 4

As described above, no disposal of dredged material occurred at Site A after late December 1982. In May 1984, 16 months after disposal operations had been halted, the site was investigated by divers using Nikonos underwater cameras to document observations. The changes in the substratum from a clay boulder, silt, and sand unpopulated bottom 16 months earlier were dramatic.

The substrate was dominated by luxuriant growths of red and brown algae. Sponges, tunicates, and sea urchins were present at all stations. The clay boulders were less densely colonized, but contained most of the species seen on rocky surfaces. All areas surrounding the rocks and boulders consisted of flat sand and shell bottoms; all silt had been swept away by natural current action. In addition, fishes were abundant at all stations.

A similar recovery has occurred since the cessation of disposal operations at Site 4. The Tampa Harbor Project was completed in late November 1985; the site was investigated by divers in July 1986, eight months after the last disposal of dredged material at Site 4. The divers found that the disposal mound at Site 4 had been heavily colonized by algae, and epifauna such as hydroids, bryozoans, ascidians, sea urchins, and arrow crabs. The sea urchins and arrow crabs

were particularly abundant on the mound. In addition, a diverse fish assemblage, including angelfishes, groupers, jacks, snappers, and wrasses, was associated with the broken relief habitat provided by boulders of dredged material on the seafloor. Most of the fishes seen are preferentially associated with hard bottom or coral reef environments.

In conclusion, it is apparent that the dredged material disposed of at both Site A and Site 4 has provided ample surfaces for colonization by numerous sessile organisms, in the same manner as artificial reefs. The dredged material also provided superb habitats and protective cover in the interstices of the rocks and clay boulders for teleosts and motile invertebrate species. The rapid recovery of Site A within 16 months, and Site 4 within eight months, to a state approaching the natural habitat of the West Florida Shelf, has provided convincing evidence that over the next several years complete recovery of both Site A and Site 4 as a coral, sponge, and fish habitat is highly probable.

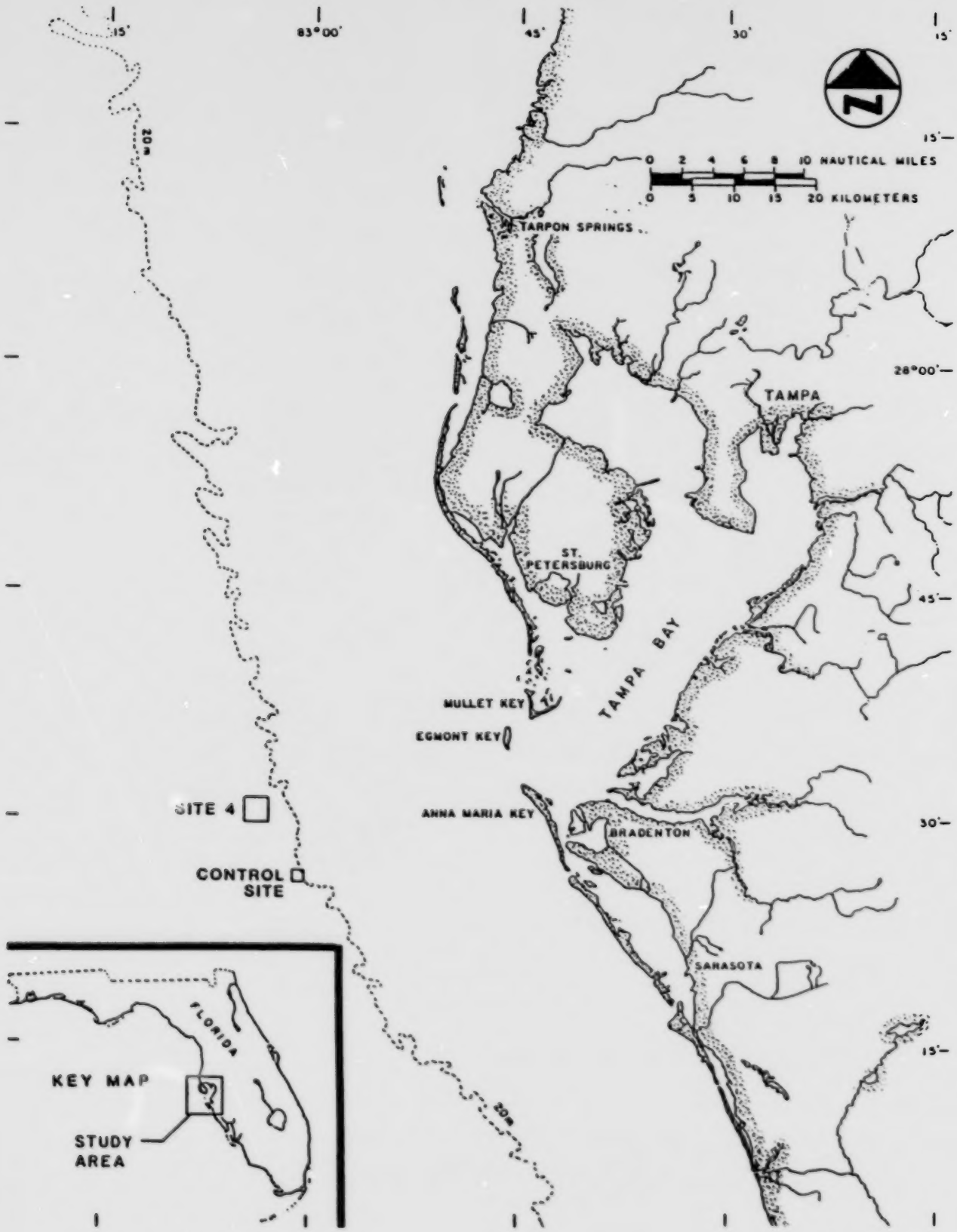

Figure 1. Location of Site 4 and the Reference (Control) Site.

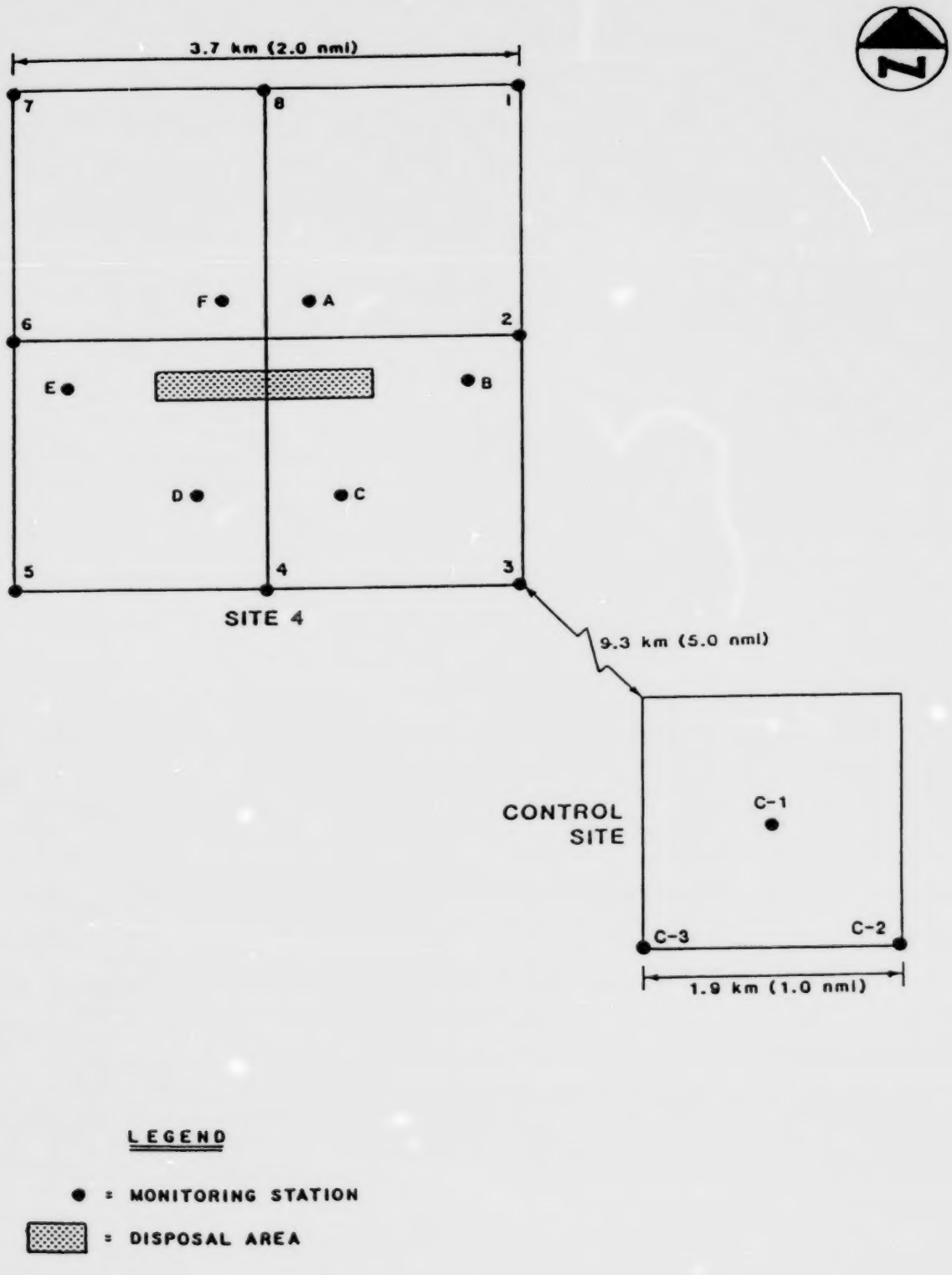

Figure 2. Monitoring stations established at Site 4 and the Reference (Control) Site.

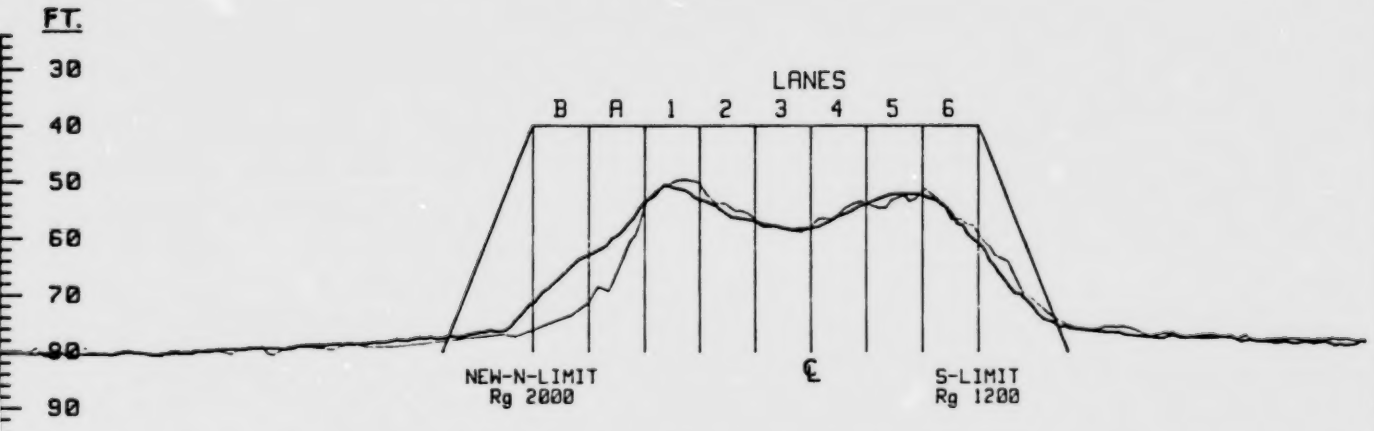

Figure 3. Cross-section of the dredged material mound at Site 4.

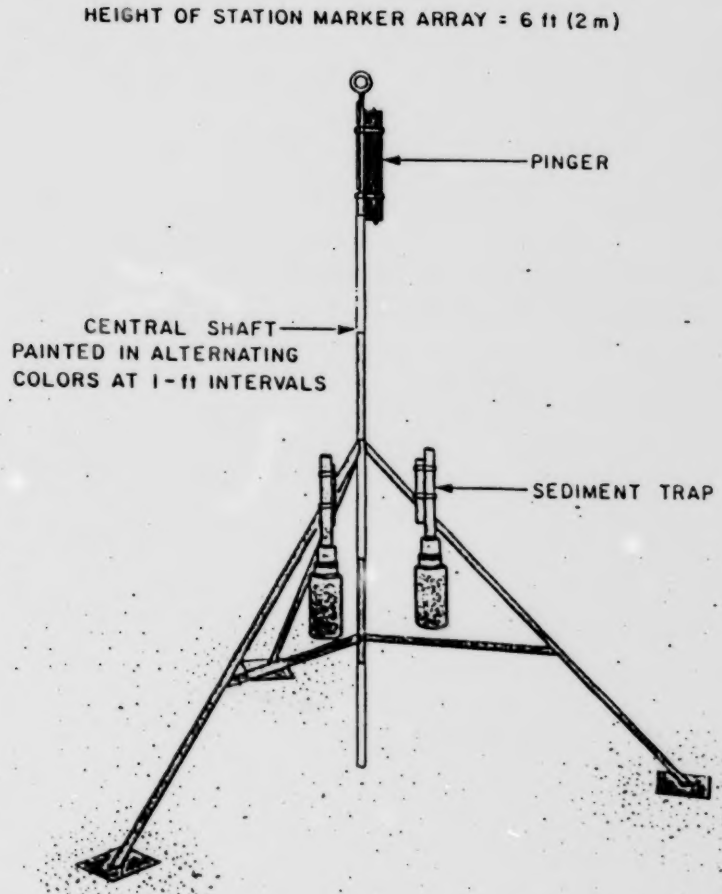

Figure 4. Station marker tripod with sediment traps and pinger.

AN OVERVIEW OF DREDGED MATERIAL OCEAN DUMP SITE MONITORING AND ASSESSMENT PROGRAMS AND DUMP SITE DESIGNATION CONDUCTED BY ENVIRONMENT CANADA UNDER THE OCEAN DUMPING CONTROL ACT IN THE ATLANTIC REGION

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ABSTRACT

This discussion paper will review the dredged material ocean dump site monitoring and assessment programs and dump site designation conducted by Environment Canada under the Ocean Dumping Control Act (ODCA) in the Atlantic Region. Ocean dump site monitoring and assessment, carried out in Nova Scotia, New Brunswick and Prince Edward Island, are described.

A study carried out at the Heron Island dump site at Dalhousie Harbour, New Brunswick, to determine the need of moving the dump site which was established in 1973 to a bathymetric depression lying adjacent to the dump site is presented. The methods used and the results and conclusions of the project are discussed.

The results of a monitoring program conducted at the dredged spoils capping site at Georgetown, Prince Edward Island are presented by comparing the bathymetric data obtained before, during and after the capping operation to assess the accuracy and the effectiveness of the capping project.

Finally, the factors considered in the selection of a dump site in the Strait of Canso, Nova Scotia, are reviewed. Chemical, physical and biological data collected during the dump site selection program and the post-dumping assessment conducted in 1985 and 1987 are discussed. Methods used in the field program included the use of a submersible vehicle and a sonar sub-bottom profiler.

Evaluation and recommendations are made on the effectiveness of these programs in relation to the designation of ocean dump sites and management decision-making under the Ocean Dumping Program in the Atlantic Region.

INTRODUCTION

Ocean dumping is regulated in Canada under the Ocean Dumping

Control Act (ODCA) which is administrated by the Environmental Protection, Conservation and Protection, Environment Canada. Approximately 125 ocean dumping permit applications are reviewed and processed each year in the Atlantic Region. As much as 90 - 95% of these applications relate to the disposal of bottom material originating from the dredging of harbours and channels of coastal ports. The Regional Director of Environment Protection is supported in the review and evaluation of the permit applications by the Regional Ocean Dumping Advisory Committee (RODAC), representing the management and scientific aspects of Environment Canada (EC) and the Department of Fisheries and Oceans (DFO). RODAC advises on the acceptability of materials proposed for dumping, the method and timing of disposal, the environmental and sociological implications of the disposal, the suitability of the dump site selected and the selection of common user dump sites. In order to correctly interpret the chemical and physical analyses of the materials proposed for dumping and to ensure that the best option of dumping methods and the most suitable disposal site are selected, a good knowledge of the actual impacts of dumping at recommended dump sites is needed. Within the existing constraints of manpower and resources, common user dump site selection and pre-dumping and post-dumping monitoring of sites considered to be most likely to have measurable impacts have been conducted either by Environment Canada or required of the permittee as a condition of the permit under the Ocean Dumping Control Act (Table 1). This paper will review three of the monitoring and assessment programs and a dump site designation conducted by Environment Canada in the Atlantic Region over the last five years.

ENVIRONMENTAL MONITORING AT THE HERON ISLAND DUMP SITE IN THE BAY OF CHALEUR NEAR DALHOUSIE, NEW BRUNSWICK

Background

In 1983, a dump site assessment program was conducted by Environment Canada at the Heron Island Dump site which is located about 10nM east of the Port of Dalhousie, New Brunswick (Figure 1). The purpose of this study was to assess the impacts of dredged spoils following the disposal of bottom material originating from the dredging of Dalhousie Harbour, the stability of the dump site and to study the physical characteristic of a nearby bathymetric depression or "trench" to determine whether or not this would be a more suitable dump site than the original site.

Methods and Materials

The field sampling was conducted by Environmental Protection personnel on July 28-30, 1981 aboard a chartered scallop dragger. Seventeen (17) sampling stations were established

Table 1. Dump Site Monitoring Conducted Under ODCA and Associated Activities.

DUMPSITE MONITORING PROGRAMS EXECUTED UNDER ODCA AND ASSOCIATED ACTIVITIES

NAME	LOCATION	TYPE OF OPERATION	TYPE OF MONITORING	REPORT YEAR	Y/N
Bergland	Glace Bay, N.S.	Vessel disposal	Sounder survey	1981	N
Canso	Strait of Canso, N.S.	Common user dumpsite	Chemical, Physical Biological	1983	Y
Canso	Strait of Canso N. S.	Common user dumpsite	Chemical, Physical, Biological, Video	1985	Y
Canso	Strait of Canso N.S.	Common user dumpsite	Chemical, Physical, Biological, Video, Bathymetric	1987	Y
Dalhousie	Heron Island, N. B.	Dredge/ Dump	Chemical, Physical, Biological, Underwater Photo	1978/ 1980	Y
Dalhousie	Heron Island, N. B.	Dredge/ Dump	Chemical, Physical, Biological, Sediment Analysis, Oceanographic	1981/ 1982	Y
Dalhousie	Heron Island N. B.	Dredge/ Dump	Sounder survey	1982	Y
Ferguson's Cove	Halifax, N. S.	Experimental Dredge/ Dump	Biological, Bathymetric, Sediment partical size	1982	Y
Fogo Island	Fogo Island, Newfoundland	Fish offal disposal	Chemical, Physical, Biological, Underwater Photo	1985	Y
Georgetown	Georgetown, P.E.I.	Capping (Dredge/ Dump)	Chemical, Physical, Biological, Bathymetric	1985	Y
Georgetown	Georgetown, P.E.I.	Capping (Dredge/ Dump)	Chemical, Physical, Biological, Bathymetric Underwater Photo	1985	Y
Lunenburg	Lunenburg, N.S.	Dredge/ Dump	Chemical, Physical Biological, Video Record, Underwater Photo	1984	Y
Lunenburg	Lunenburg, N. S.	Dredge/ Dump	Chemical, Video Record	1985	Y
Miramichi	Miramichi River N. B.	Dredge/ Dump	Chemical, Physical Biological, Oceanography, Bathymetric Aerial Photo	1979/ 1981	Y
"Ocean Ranger"	Flemish Pass	MODU disposal	Video Record and sounder survey	1984	Y
Pictou Hr.	Pictou, N.S.	Dredge/ Dump	Physical, Biological, Oceanographic, Bathymetric	1987	
Pigeon Hill	Pigeon Hill, N.B.	Dredge/ Dump	Physical, Biological, Oceanographic	1987	
Saint John Hr.	Saint John, N.B.	Dredge/ Dump	Chemical, Biological	1981	
Saint John Hr.	Saint John, N.B.	Dredge/ Dump	Chemical, Biological	1982	
Saint John Hr.	Saint John, N.B.	Dredge/ Dump	Chemical, Biological, Oceanographic	1983	Y
Yarmouth	Yarmouth, N.S.	Herring Wastes	Physical, Video	1987	

Figure 1. Heron Island Dump Site.

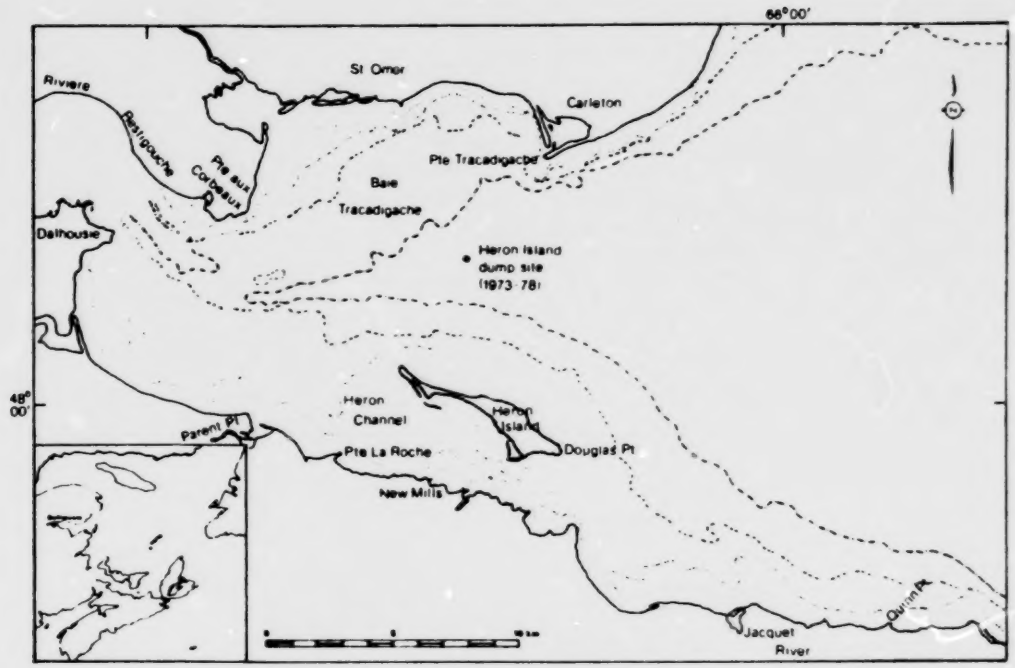


Figure 2. Sample Stations.

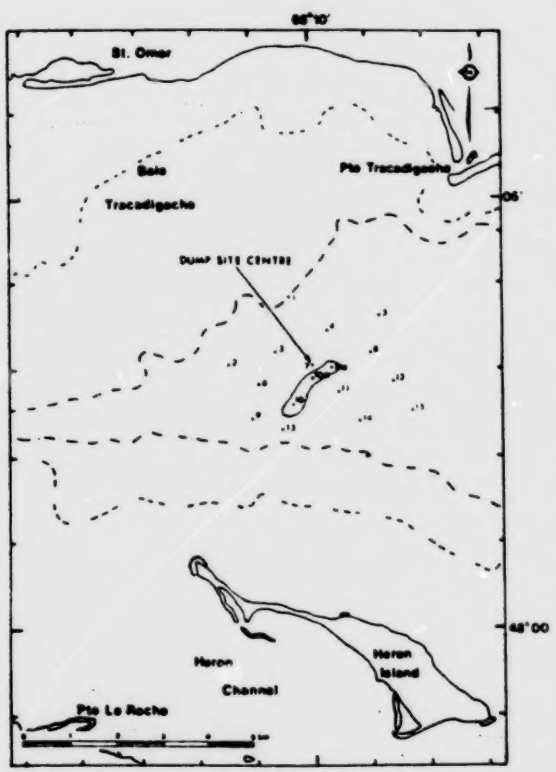
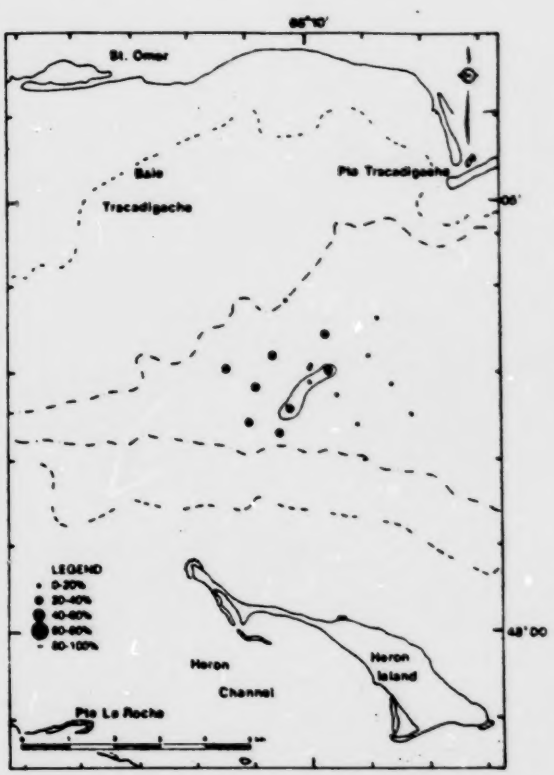


Figure 3. Distribution of Silt and Clay.



using horizontal sextant angles and the trench was located using a chart recording depth sounder (Figure 2). A Peterson grab sampler (0.1 m2 sampling area) was used to collect sediment and biota samples. A 0.5 mm screen was used to collect the benthic infauna. Current data were obtained by Anderraa current meters at the dump site center and the trench from November 5 to December 9, 1981.

Results and Discussion

Sediment size study

The high proportion of coarse material in the sediment suggest that a certain amount of sediment re-suspension and sorting may occur in some portions of the study area. Figure 3 indicates a trend towards increasing percentages of silt and clay in the west and south west portions of the sample grid. These observations are supported by the data of Schafer and Bezanson (1980) which demonstrate a tendency for sea bed drifters deployed at the dump site to be recovered in areas south west and west of the dump site. Because of the high gravel and wood chips found in the trench, it is apparent that some material had already been dumped directly in the trench area and also redistributed to that location by natural processes. It is also quite possible that finer material fractions in the trench were subject to resuspension and redistribution.

Trace metals

Figure 4 shows that trace metal concentrations of sediment samples appear to be higher at the dump site centre and to the west of the dump site which corresponds with indications from other aspects of the study that re-distribution of dredge spoils in a south westerly and westerly direction may be occurring.

Benthic Infauna

The infaunal densities ranged from a mean of 123.3 individuals/0.1 m2 at Station 2 to a mean of 8.3 individuals /0.1 m2 at Station 7 where the most pronounced effects of disposal were observed (Table 2). The dominant species over the sample grid were the bivalve Thyasira flexuosa; the polychaetes Maldane sarsi, Tharyx acutus, Scolopolos armiger, and the brittle star Ophiura sarsi.

Figure 5 shows plot of the cumulative percentage of species against the geometric class of individuals per species on probability paper. The bend in the log-normal distribution represents a departure from an equilibrium community (Gray and Mirza 1979; Gray 1981). Under slight pollution or other external stresses, the equilibrium community is disturbed, and some species become more abundant, whereas the rare

Figure 4. Trace Metals in Sediment: (A) Cd, (B) Pb, (C) Hg, (D) Cu.

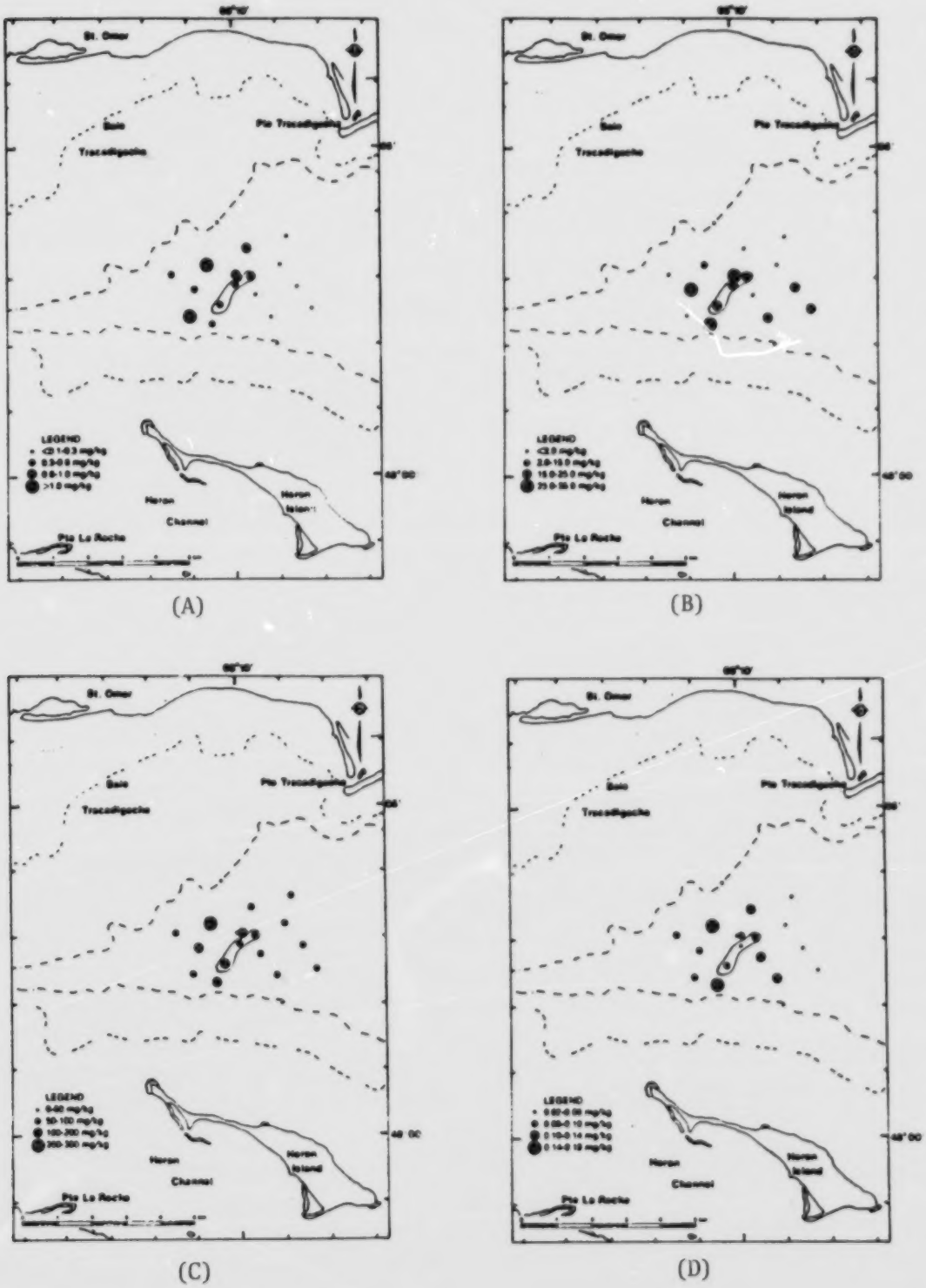
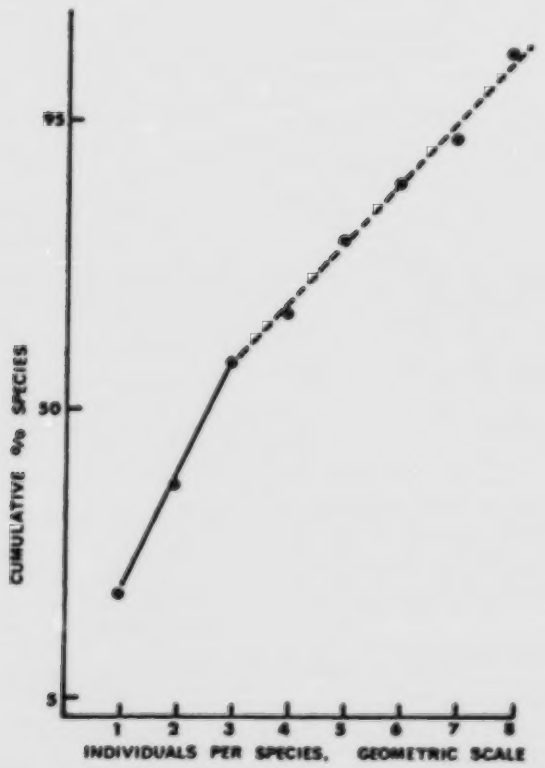


Table 2. Benthic Infauna From Study Area

	TOTAL NUMBER OF INDIVIDUALS					
	REPLICATE 1	REPLICATE 2	REPLICATE 3	MEAN	S.D.	S.E.
1	20	15	22	19	2.94	1.68
2	114	123	133	123.3	7.76	4.48
3	37	50	51	46	6.38	3.68
4	71	47	16	44.7	22.5	12.99
5	109	112	69	96.7	19.6	11.32
6	31	27	72	43.3	20.34	11.74
7	24	0	1	8.3	11.09	6.40
8	54	38	55	49	7.79	4.5
9	58	42	54	51.3	6.8	3.93
10	38	88	52	59.3	21.06	12.16
10A	5	43	47	31.7	18.93	10.93
11	134	109	102	115	13.74	7.93
11A	54	46	56	52	4.32	2.49
12	78	101	121	100	17.57	10.14
13	74	50	39	54.3	14.61	8.44
14	77	83	76	78.7	3.09	1.78
15	87	95	75	85.7	8.22	4.75

Figure 5. Log-Normal Plot of Benthic Infauna From Study Area.



species do not change abundance. Thus the first part of the log-normal curve follows a straight line log-normal pattern, whereas from geometric class 4 the line has a shallow slope extending over the rest of the geometric classes.

Current Measurements

The Anderraa current meters deployed at the Heron Island dump site and adjacent trench site recorded temperature, salinity and current speed and directions at 1.3 m above the bottom from November 5 to December 9.

Data obtained by the current meters indicated maximum current speeds at the existing Heron Island disposal site of 39.68 cm/sec setting towards 263 T at 1.3 meters above the bottom while at the trench mooring site the maximum speed recorded at 1.3 meters above the bottom was 35.48 cm/sec setting towards 317 T. The minimum current speed recorded at both sites was 1.5 cm/sec; however, the threshold of the meters is limited to 1 cm/sec. The data indicate that the current regimes at the existing Heron Island Dump Site and at the adjacent trench site are very similar.

Conclusion

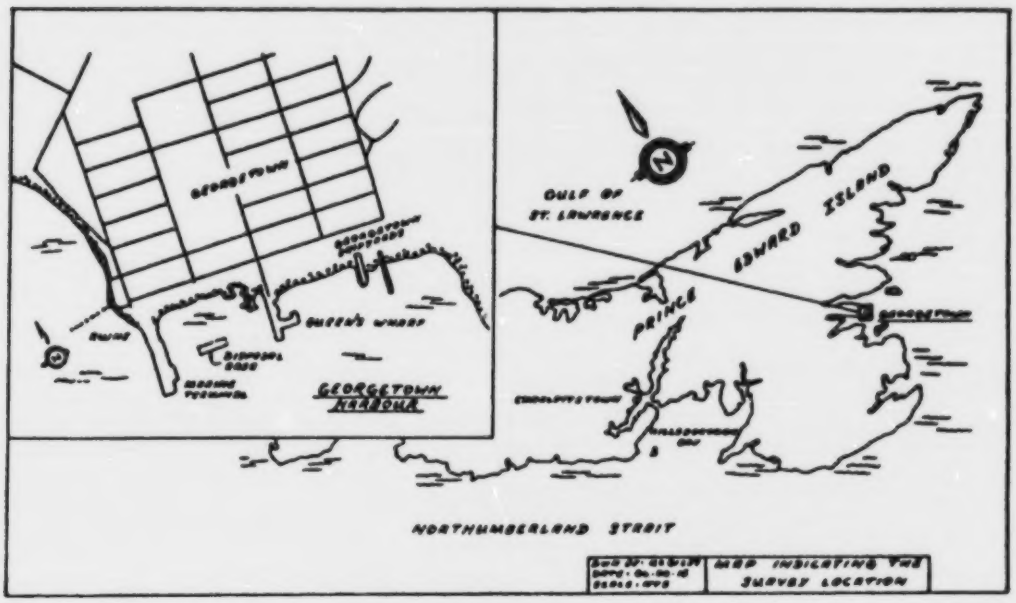
The results of this study indicated that elevated levels of trace metals and low density of infauna were found at the center of the Heron Island Disposal site where pronounced effects of disposals occurred. Since the current measurements showed that the current regimes at the existing dump site and the adjacent trench site are very similar, it is concluded that there is not enough justification to expand the immediate area of impact by moving the dump site to the trench.

PCBs CONTAMINATED SEDIMENT CAPPING SITE AT GEORGETOWN HARBOUR, PRINCE EDWARD ISLAND

Background

The practice of covering somewhat contaminated spoils with clean material often referred to as capping so that it is isolated and thus ensuring that it is rendered harmless to the marine environment has been documented by a number of researchers (Bokuniewicz et. al. 1983; Bokuniewicz 1985; Brannon et. al. 1984; Gunnison 1984). In 1985, the first capping monitoring program in the Atlantic Region was conducted at the request of Environment Canada by Public Works Canada at Georgetown Harbour which is located at the east coast of Prince Edward Island (Figure 6). Approximately 6,000 cubic meters of dredged material originating from the Public Wharf contains levels of PCBs up to 4.0ppm were placed in a pit excavated in the sea floor in an adjacent area and

Figure 6. Location of Georgetown, Prince Edward Island (OceanChem, 1986).



covered with a minimum of 1.0 meter of clean material. The work was closely monitored by Environment Canada using bathymetric measurements as a means of ensuring that an adequate cap was established.

Methods

Sounding data of the capping site were obtained before and after the excavation, immediately after the contaminated material was dumped into the excavation, immediately after the contaminated material was capped and six months after the capping operation. All sounding survey was conducted by the sounding crew of Public Works Canada. Range finders with an error of ± one meter were used for positioning.

An IBM PC was used to manipulate the sounding data before they were plotted using the mainframe computer of Bedford Institute of Oceanography.

The excavation was conducted by a clam dredge floating plant and the dumping and capping operations were carried out by a barge and a sectional scow.

Results and conclusion

Figure 7 shows a series of 3-D profiles of the capping process. Based on the pre-capping and post-capping sounding data, the average thickness of the final cap was calculated as 1.01 - 1.31 metres. The effectiveness of the cap was confirmed by a sediment coring program at the capping site.

The results of this monitoring program indicated that capping, if done properly, could be a very effective option for the disposal of contaminated material. Experience gained from this monitoring program will assist RODAC in the planning of any future capping in the Atlantic Region. At the present time, a laboratory research project to study the rate of migration of PCBs within the sediments and cap is being carried out at the Bedford Institute of Oceanography to further study the effectiveness of this type of capping project.

CANSO COMMON USER DUMP SITE SELECTION AND PRE-DUMPING AND POST-DUMPING MONITORING PROGRAMS

Background

The Strait of Canso is a narrow body of water connecting the St. George Bay and the Chedabucto Bay on the east coast of mainland Nova Scotia, west of the Cape Breton Island. In 1954, a causeway was constructed across the strait joining Mainland Nova Scotia and Cape Breton. Since the construction of the causeway, the part of the strait south of the causeway had attracted many industrial developers because of its ice-free deep water.

In 1983, due to the offshore hydrocarbon exploration and the potential for development of the Venture Field in this area, and due to the need of harbour development required by the existing industrial sites within the strait, a decision was made by RODAC to identify a common user dump site for this area.

Canso Dump Site Characterization and designation

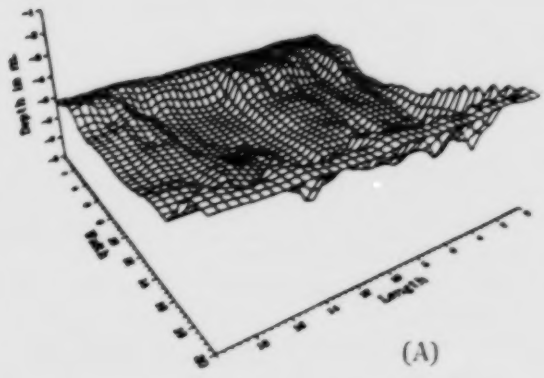
RODAC identified the following reasons for designation of a common dump site in the Strait of Canso (Martec 1984):

a. Forestalling a proliferation of smaller dump sites so that the area impacted by dumping is fairly small and the impacts are concentrated;

b. Formal designation of a common dump site will allow future proponents to develop accurate budgets for projects involving dredging and spoil disposal;

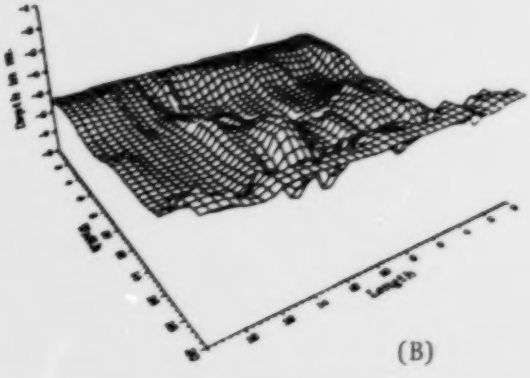
Figure 7. 3-D Profiles of the Capping Process:
(A) Post-Excavation, (B) Post-Dumping,
(C) Immediately after Capping, (D) Six Months after Capping.

Georgetown Post-Excavation



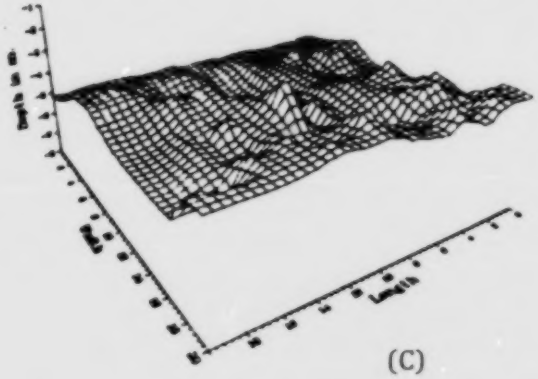
(A)

Georgetown Post-Dumping



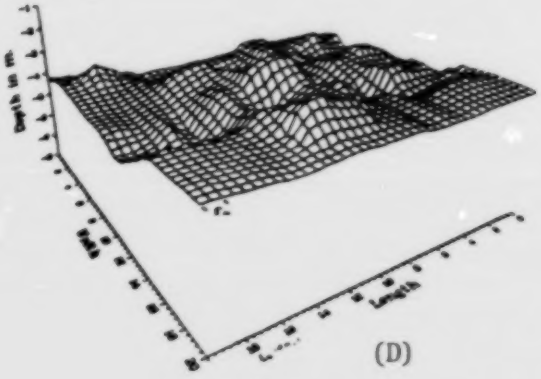
(B)

Georgetown Post-Capping



(C)

Georgetown 1986 Soundings



(D)

c. Reducing the time and costs to both proponents and regulatory agencies presently required to designate dump sites for each permit application;

d. Making the monitoring of the effects of dumping in the Strait of Canso region more feasible.

Based on the above justifications, a resources map was prepared by DFO to identify the prospective dump sites for this region. Four sites were eventually selected (Figure 8). Information relevant to these dump sites from a literature search was compiled. To supplement the existing information, a field program was initiated by Environment Canada and conducted by both EPS and Martec Ltd. at these sites to collected physical, biological and chemical data. A ranking system and the criteria used for the dump site selection were developed (Tables 3). Based on the ranking system, the site located immediately north of Pirate Harbour in the Strait was selected as the most favorable dump site since it had the least benthic activity of the other sites and the highest concentration of sediment contaminants. It was thus concluded that this site should be recommended to RODAC as the Canso Common User Dump Site because dumping of dredging spoils at this site would have the least environmental and socioeconomic impacts (Martec 1984).

Post-dumping Monitoring Programs at the Canso Common User Dump Site

In late 1983, the site selected by the dump site selection study was designated by RODAC for the disposal of uncontaminated dredge spoils from all the dredging operations within the Strait of Canso. To date, a total of 57,000 cubic meters of dredge materials (54,000 cu.m. from Mulgrave Harbour from October to December 1983; 3,700 cu.m. from Port Hawkesbury in March 1984) have been disposed of at this site.

1985 Post-dumping Monitoring Program

During the period of July 16 - 19, 1985, a field program was conducted by Environmental Protection Service personnel to study the impacts of dumping at the dump site (Tay 1987). A remote control underwater submersible, the "DART", was used with the assistance of the DND personnel to obtain a visual record of bottom characteristics of the site.

Sediment samples and benthic infauna samples were collected using a 0.05 m2 Ponar grab at thirteen sampling stations (Fig. 9). Positioning was established using a Loran C aboard the Bedford Institute of Oceanography research vessel "NAVICULA". Triplicate benthic infauna samples were collected at each station. A 0.5mm screen was used for sieving. All organisms were fixed in a 10% buffered saline-formalin

Figure 8. Prospective Dump Sites Selected For Ranking Study (Martec, 1984): SC1, SC2, SC3, SC4.

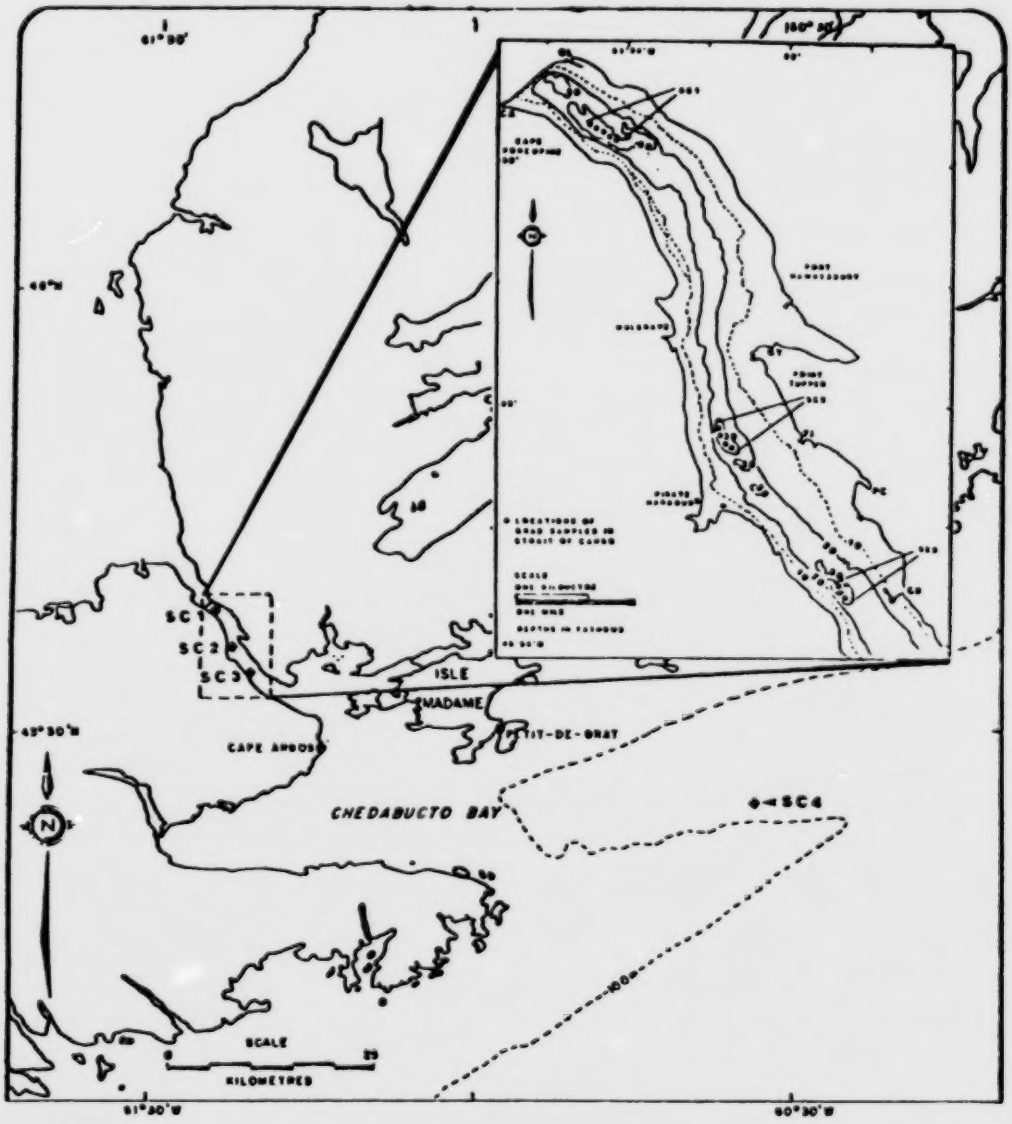


Table 3. Ranking System and Criteria Used for Dump Site Selection (Martec, 1984).

RANKING OF FOUR PROSPECTIVE DUMP SITES IN
THE STRAIT OF CANSO/CHEDABUCTO BAY REGION -
CATEGORIES IN WHICH LOW RANKING IS FAVOURABLE TO SPOIL DISPOSAL*

Category	SC1	SC2	SC3	SC4	
Water Depth	3	1	2	4	
Tidal Current Velocity	1	2	3	4	
Extreme Current Velocity	1	2	3	4	
Number of Faunal Taxa in Grabs	1	3	2	4	
Density of Grab Fauna	2	1	3	4	
Biomass of Grab Fauna	2	1	3	4	
Density of Bivalves	1	1	1	2	
Density of Polychaetes	2	1	3	4	
Density of Amphipods	1	2	2	3	
Density of Echinoderms	1	1	2	3	
Number of Faunal Taxa in Trawls	2	1	2	-	**(3)
Density of Trawl Fauna	3	1	2	-	(4)
Frequency of Occurrence of Shrimp	3	1	2	-	(4)
Frequency of Occurrence of Echinoderms	2	1	3	-	(4)
Frequency of Occurrence of Fishes	2	3	1	-	(4)
Trap and Net Fishery	4	2	3	1	
Groundfishery	1	1	1	2	
Vessel Traffic	2	3	4	1	
Distance from Main Potential Dredging Sites	3	1	2	4	
Total	37	29	44	63	

* 1 = lowest value in a given category
4 = highest value in a given category
similar values are ranked the same

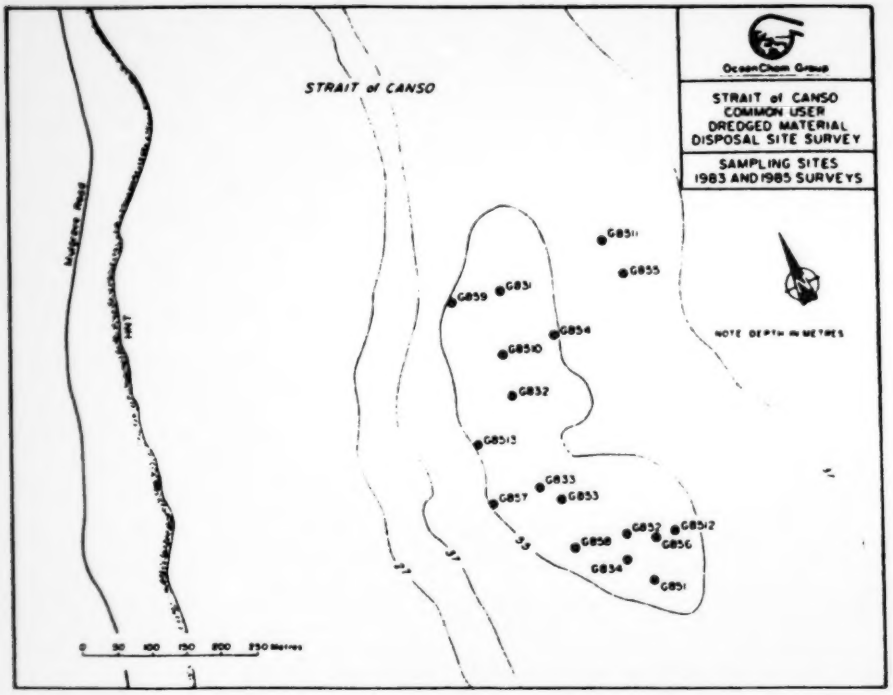
** - no data, value assumed, based on grab data

RANKING OF FOUR PROSPECTIVE DUMP SITES IN
THE STRAIT OF CANSO/CHEDABUCTO BAY REGION -
CATEGORIES IN WHICH HIGH RANKING IS FAVOURABLE TO SPOIL DISPOSAL*

Category	SC1	SC2	SC3	SC4
Incidence of Silt/Clay in Surficial Sediments	3	4	1	2
Mercury Concentration	3	3	2	1
Lead Concentration	4	3	2	1
Zinc Concentration	4	3	2	1
Cadmium Concentration	4	3	2	1
Copper Concentration	4	3	2	1
op DDE Concentration	3	4	2	1
pp DDE Concentration	2	3	1	1
pp DDT Concentration	2	4	3	1
PCB Concentration	2	4	3	1
Distance to Industrial Water Users	3	1	2	4
Total	34	35	22	15

* 1 = lowest value in a given category
4 = highest value in a given category

Figure 9. 1985 Post-Dumping Sampling Stations (OceanChem, 1987).



solution. Sorting, weighing, enumeration and identification of the fauna were carried out in laboratory. A reference sample was kept for future reference.

The results of this study showed that the sediment particle size at the dump site had shifted from fine silt and clay towards the coarse particles of gravel and sand after the dumping operations. Concentrations of heavy metals are, in general, higher in the post-dump samples than those collected in the pre-dump survey. Eight of the thirteen post-dump sampling stations had high PCB levels (104 - 1015 ug/kg) (Table 4).

Polychaeta and Nemertinea are the most abundant groups both before and after dumping. The number of polychaetes increased after the dumping operations. Prionospio steenstrupi (44%) and Ninoe nigripes (25%) are the most dominant species among the 14 species identified. A large number of Foraminifera (2704 - 5728 per grab) was found at the stations located at the northeast end of the dump site area. The occurrence of this group of organisms at an area which has undergone substantial anthropogenic modification was reported by Schafer in his 1982 Dalhousie Heron Island dump site study (Schafer, 1982).

The visual observation of "protruding clumps" of dredged spoils at the dump site and the significant changes of the sediment particle sizes and the concentrations of some heavy metals and PCBs indicated that dumping operations had changed the natural environment of the dump site.

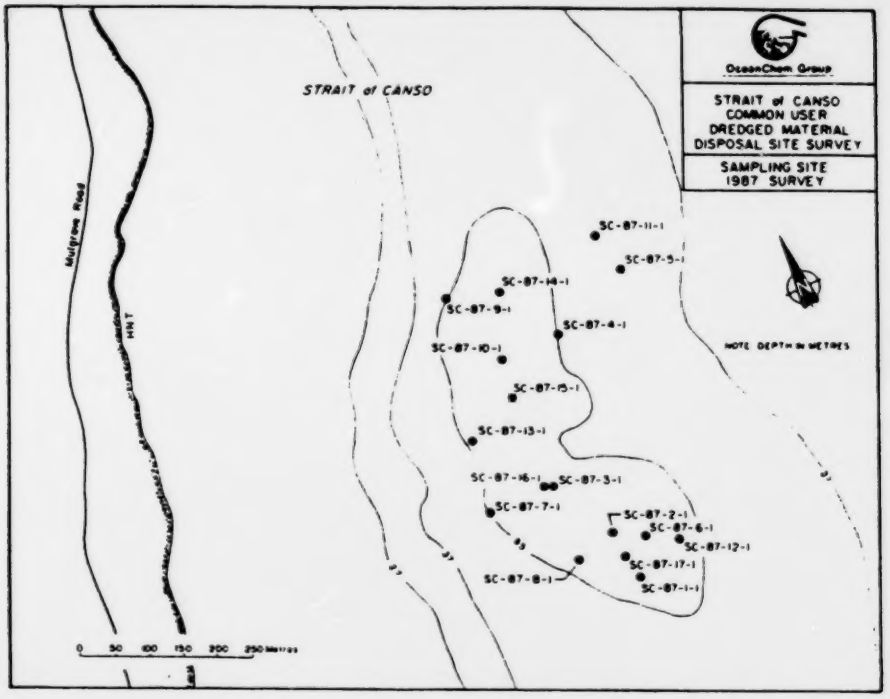
Table 4. Summary Statistics for Particle Size, Trace Metals, and PCBs of Sediment Samples Collected in 1983 Pre-Dumping and 1985 and 1987 Post-Dumping Surveys.

	1983	1985	1987
Particle size (%)	(n=4)	(n=24)	(n=17)
Gravel	0	7.9± 10.8	1.6± 3.3
Sand	4.0± 1.3	27.2± 12.9	35.3± 11.1
Silt	51.9± 2.0	40.0± 11.4	44.1± 9.1
Clay	44.1± 1.4	26.5± 9.4	19.0± 4.7
Trace metals (ppm)			
Cd	0.45±0.17	0.65±0.16	0.54±0.11
Zn	116.3± 3.4	139 ± 28	141.1± 12.6
Pb	44.5± 0.5	58.9± 14.2	48.4± 21.2
Cu	37 ± 5	32.0± 17.8	103.1±113.1
PCBs (ppb)	80 ± 4	167.2±216.8	157.8±101.1

1987 post-dumping monitoring program

Post-dumping monitoring program at the Canso dump site was continued on July 20 -23, 1987. Physical, chemical and biological data from 17 stations corresponds to stations sampled in the 1983 pre-dumping and 1985 post-dumping surveys were collected by Environment Canada personnel and OceanChem Group (Figure 10). Sediment samples were also collected from 17 stations selected randomly outside the dump site. The program was designed to study the recolonization of the dump site and the spatial distribution of PCBs and Cd in the dump site and its surrounding areas.

Figure 10. 1987 Post-Dumping Sampling Stations (OceanChem, 1987).

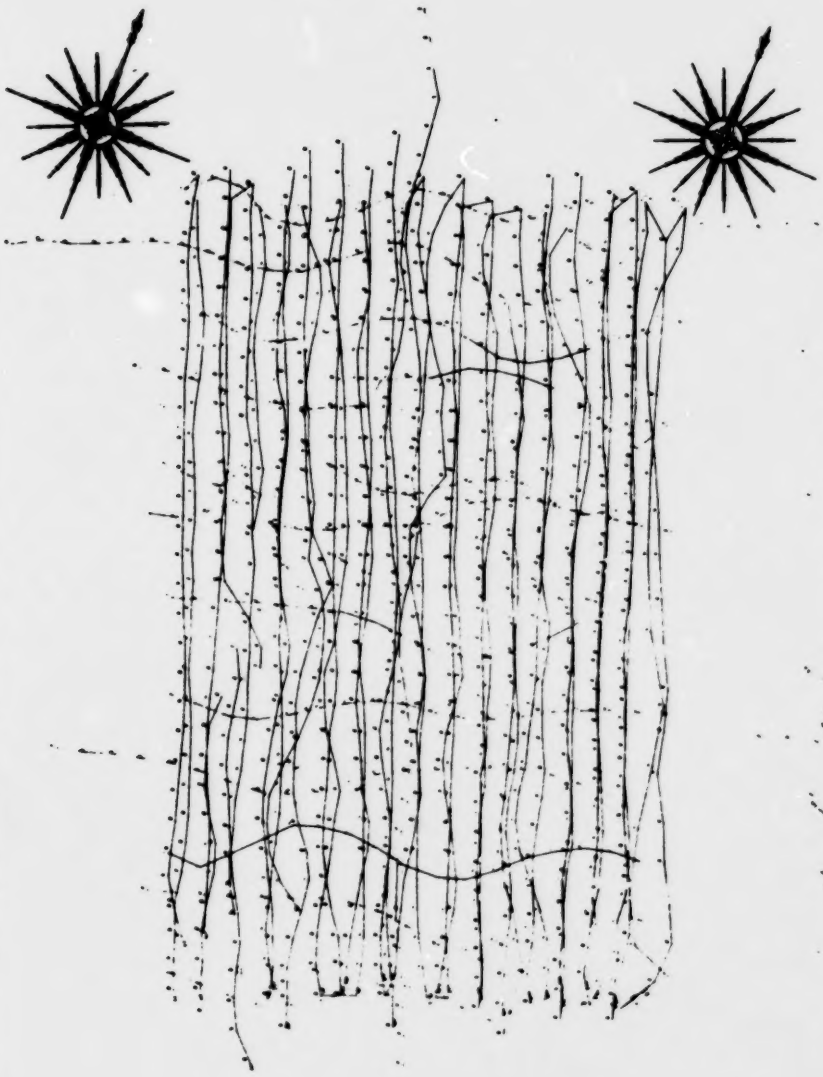


Before the monitoring program was carried out, a geophysical survey was conducted by McElhanney Services Ltd. to obtain bathymetric data and sub-bottom profile at the dump site. Geophysical data were collected over a rectangular survey grid comprised of 34 NNW - SSE oriented lines, spaced approximately 10 meters apart, and 12 ENE - WSW oriented lines, spaced approximately 50 m apart (Figure 11A). Positioning and navigation were provided by a Del Norte Trisponder System integrated to McElhanney's computer navigation package "NAVPAK".

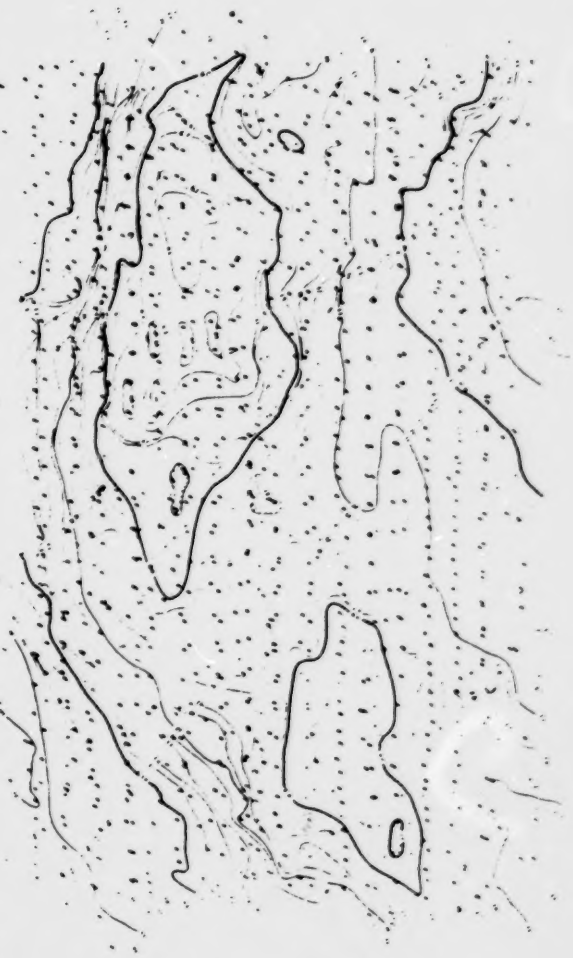
The site specific bathymetry indicated that depth over the bottom of the dump site varied between 45 m to 65 m. As shown in the bathymetry map (Figure 11B), the bottom forms a depression in the centre of the site with a ridge running north to south throughout the site. The sub-bottom profiling showed a homogeneous sediment composition throughout the area. The exception to this is a surficial veneer (0 - 2 m) of seismic reflections that is too inconsistent to be mapped (McElhanney 1987).

Sediment and biota samples were collected from the 17 stations within the dump site using a 0.25 m2 Van Veen Grab. Triplicated infauna samples were collected from the Van Veen

Figure 11. (A) Geophysical Survey Grid,
(B) Bathymetric Data of the Study Area (McElhanney, 1987).



(A)



(B)

sample using a 0.05 m2 Ponar grab to maintain consistency in sampling methods used previously. All stations were located by the Trisponder System used by McElhanney. The error in distance from the 1983 and 1985 sites ranged from two meters to 35 meters. Sampling, sieving and preserving of samples were similar to the 1985 monitoring program.

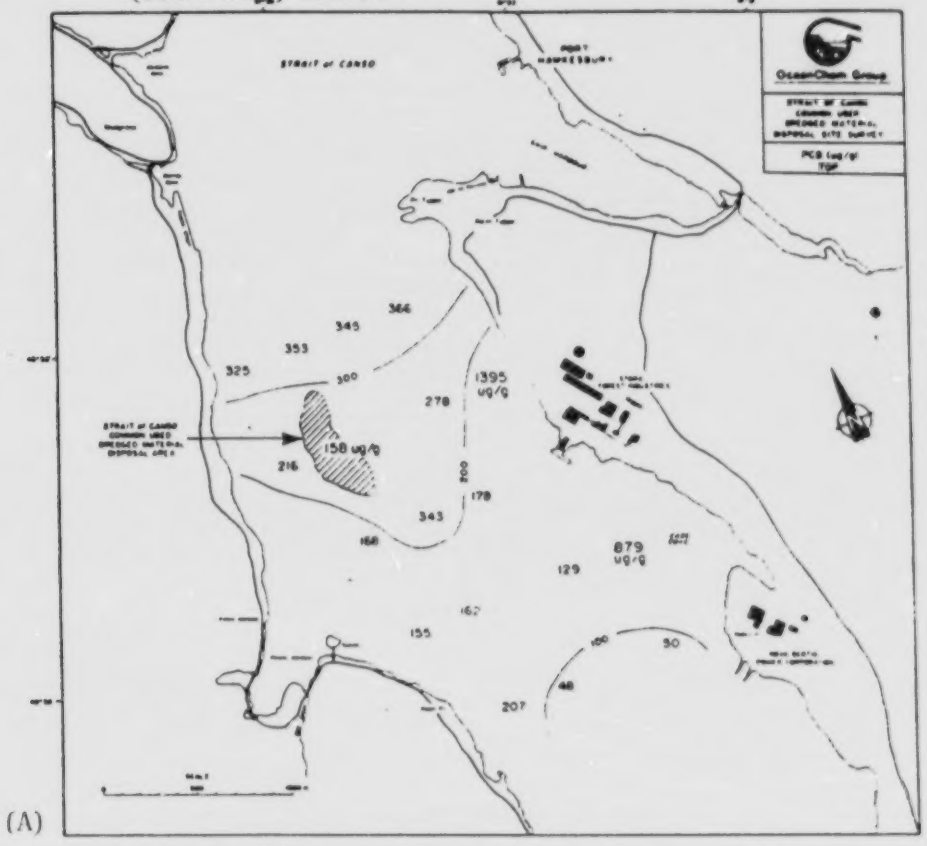
Sediment particle size analysis indicated that there is an increase of fine silt and clay on the surface sediment at the dump site. However, the results are, in general, similar to those reported in the 1985 study (Table 4).

The levels of trace metals did not change significantly from 1985 to 1987 except copper which had increased significantly in the 1987 survey (Table 4).

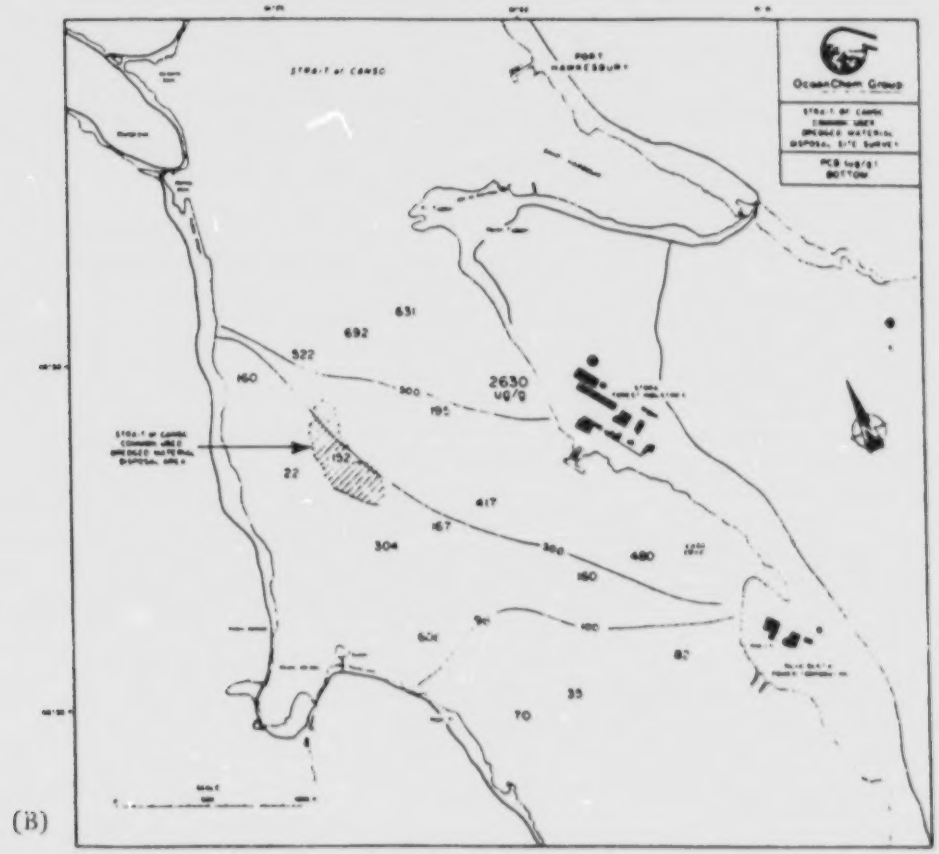
The 1987 PCBs levels are higher than the pre-dump survey and similar to those obtained in 1985 (Table 4). However, high levels of PCBs were also detected in all the other stations outside of the dumping area (Figure 12). Comparison of the GC traces of PCBs measurements obtained from this survey with those obtained from a potential dredged site located at the Nova Scotia Power Corporation 2 nM (2,000 meters) southeast of the dump site indicated that the PCBs could have originated from the same source. The GC traces also show significant amount of unknown organic chemical compounds existing in the sediments (MacKnight, pers. comm.). No further attempts were carried out to identify these compounds.

Both the number of species and individuals of Polychaetes had increased significantly from 1985 to 1987. Prionospio steenstrupi, a second stage recolonizer, is the dominant species both in 1985 (44%) and 1987 (36%). Other second stage recolonizer collected in 1987 are members of the genera, Lumbrineris, Eteone, Schistomeringos, Pholoe, and Nereis (Pocklington 1987). The total number of polychaetes species increases from 14 in 1985 to 37 in 1987 while the total number of individuals increases from 625 to 2530. The increase of number of species and individuals of polychaetes and the domination of the second stage recolonizer found in the 1987 sediment samples indicated that the infauna community at the dump site is undergoing a recolonization process. Log-normal distributions of the infauna data of the 1985 and 1987 surveys are illustrated in Figure 13. A break was shown in both the 1985 and 1987 data indicating a departure from an equilibrium community. It is not known that the disturbance was caused by the dumping operation or other external stresses. However, the break in the 1987 data is less distinct and the data stretch over 9 geometric classes, three more than the 1985 data. The better fit of the 1987 data to the log-normal distribution means the infaunal structure of the community is recovering from the original impact.

Figure 12. Distribution of Sediment PCBs at Canso Dump Site and Surrounding Areas: (A) Surface (B) 0.3m From Surface (OceanChem, 1987).

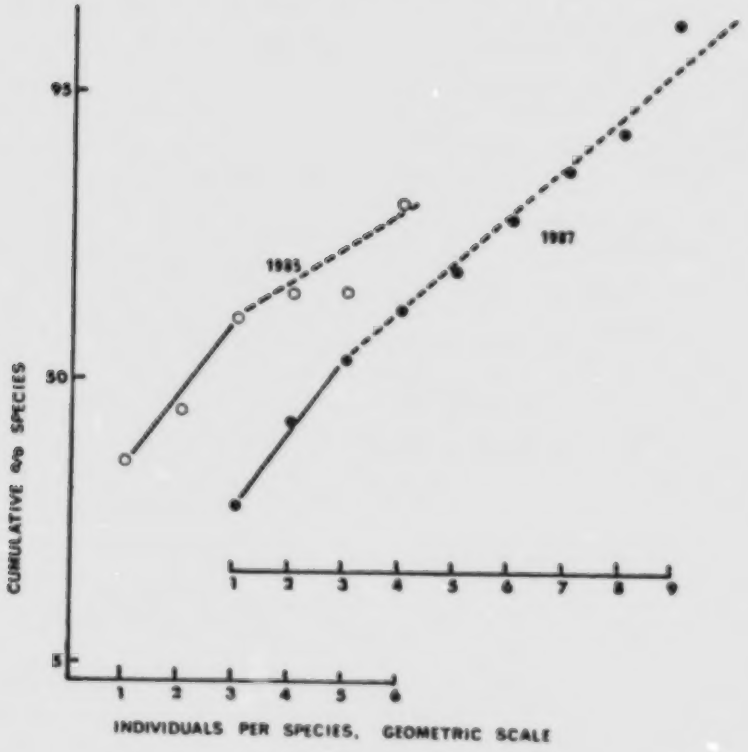


(A)



(B)

Figure 13. Log-Normal Plots of Benthic Infauna Collected in the 1985 and 1987 Surveys.



CONCLUSION

The three examples presented above are representatives of the types of monitoring programs which have been carried out in the Atlantic Region under the auspices of Ocean Dumping Control Act. The results of these assessments have provided useful information to the Atlantic Regional Ocean Dumping Advisory Committee in the validation of their recommendation and decision making process.

The majority of these monitoring programs were conducted under the Ocean Dumping Research Fund the national total of which amounts to only $150K annually. This year, 80K was allocated to Atlantic Region for the following studies:

a. Transportation of contaminants through coastal marine sediments, Phase II;
b. Canso common ocean dump site, post-dumping assessment (continued);
c. Marine disposal of fish offal;
d. Development of rapid cost-effective sediment bioassay procedures based on activity of indigenous microorganisms;
e. 7th Internal Ocean Disposal Symposium.

All the above projects with the exception of the later are related to dump site assessment and monitoring. In addition, special efforts were made by RODAC to explore new, cost-effective techniques such as the use of a remote-control submersible to survey dump site and the deployment of sea-bed drifters at various designated dump sites to study sediment movement.

Dump site monitoring and assessment are important components of the Atlantic Regional Ocean Dumping Program. With the support of the Ocean Dumping Research Funds, funding by ocean dumping permittees and with the involvement of scientists and managers who are active in this field, an effective and productive program is being developed and maintained within this region to provide essential information for the scientific management of Ocean Dumping and the protection of our marine environment.

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My thanks to Mr. Andre d'Entremont for his assistance in the preparation of all the Georgetown sounding data and to Mr. Paul Mudroch, Dr. Scott Macknight and Miss Tara Marshall for their able assistance in samples collection and report preparation for the 1987 Canso study. I also want to thank Dr. Glenn Atkinson and Mr. Francis Jordan for their useful comments in sampling design for the Canso studies.

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SIMULATION OF DREDGED MATERIAL DISPERSAL DURING AQUATIC DISPOSAL

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ABSTRACT

An empirical model describes dispersal of dredged material disposal plumes in an urban estuary, Central Long Island Sound, during spring of 1983. The model uses in situ current meter data and records of disposal activities. Disposal plumes are assumed to form as sub-pycnocline clouds which have uniform distribution in the vertical and Gaussian in the horizontal. The clouds grow horizontally with a constant diffusion velocity and are transported by measured currents. Results include trajectories of individual disposal plumes and elevations in suspended solids concentrations at specific locations resulting from multiple disposal events. These simulations indicate potential far-field transport of suspended dredged material and associated contaminants.

INTRODUCTION

EPA's Environmental Research Laboratory-Narragansett (ERLN) participated in the Interagency Field Verification Program (FVP) with the U.S. Army Corps of Engineers (COE) to field verify existing predictive techniques for evaluating environmental consequences of dredged material disposal

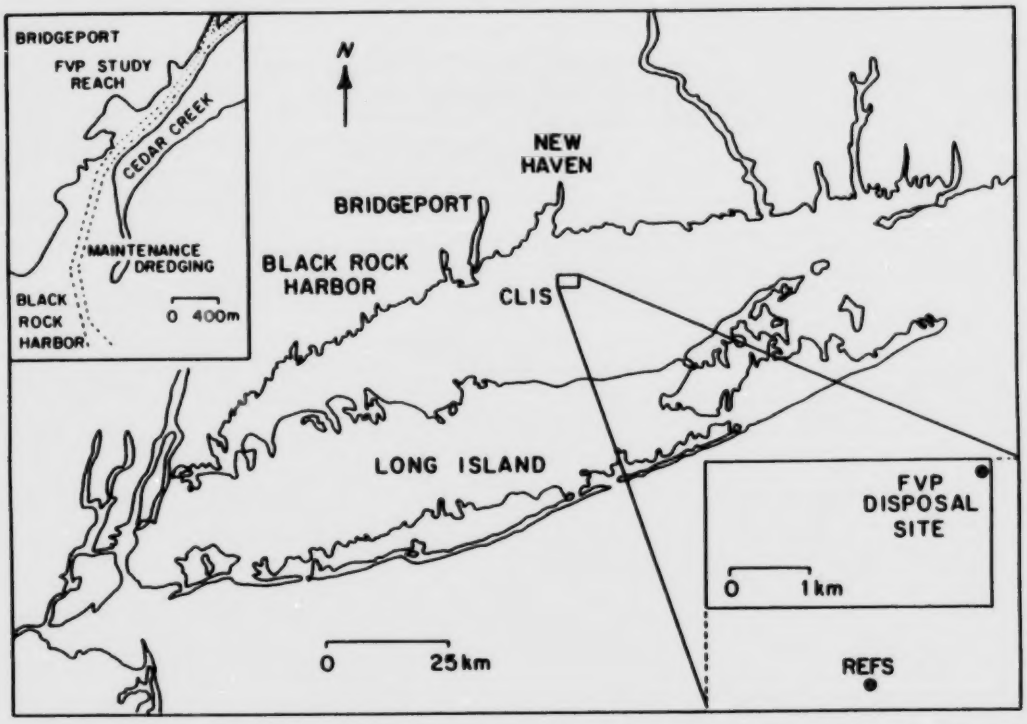

Figure 1. Long Island Sound and the locations of Black Rock Harbor, CLIS disposal area, and FVP study site.

under aquatic, intertidal, and upland conditions (Gentile and Pesch 1987). The aquatic portion of FVP was conducted by ERLN.

The FVP aquatic disposal site lies within an historical site known as the Central Long Island Sound (CLIS) Disposal Site (1.8 by 3.7 km) located approximately 15 km southeast of New Haven, Conn. (Figure 1). Water depth is approximately 20 m. The study area sediments are primarily silt-clay, of mean grain size 0.013 mm. Thermal stratification occurs from April to September, with bottom salinity slightly higher than that of the surface during this period. Tidal currents typically dominate the near-bottom water motion in a predominantly east-west direction.

The actual FVP disposal location was selected within the CLIS disposal site so as to minimize contamination from other sources, including relic disposal operations and ongoing disposal activities occurring during the study period. Stations used to examine biological effects were selected along the primary axis of tidal-current flow to represent a gradient of potential dredged material exposure of the biota.

The dredging site for the FVP was Black Rock Harbor, located in Bridgeport, Conn. (Figure 1). Approximately 55,000 m^3 were dredged during April and May 1983, and disposed in the northeastern corner of CLIS disposal site. Concurrent operations from March through June 1983 within the CLIS disposal site accounted for disposal of approximately 10^6 cubic meters of dredged material.

Two important aspects were necessary to provide a predictive link to biological effects measured in field-exposed organisms: (1) quantification of actual exposures of biota to dredged material during the study period, and (2) determination of possible interference from other ongoing disposal operations within CLIS site during the study. To address these, an empirical model was developed to describe the dispersal of disposal plumes within CLIS.

METHODS

An empirical model was developed to simulate dispersal of dredged material plumes. The plumes are assumed to form

as sub-pycnocline clouds by retention in the water column of fine particulates and dissolved materials following gravitational descent of the dredged sediments from disposal vessels. Material within each plume is assumed to have uniform distribution in the vertical dimension and to be Gaussian in the horizontal dimensions. Plumes are transported by the local current (as obtained from in situ instrumentation, described below), and grow according to a constant diffusion velocity. No loss of material out of the disposal plume is assumed during dispersal, i.e., material in the plume is completely conserved. The method therefore provides upper bound estimates for water column exposures.

From April to July 1983, the DAISY in situ instrumentation array (Bohlen and Winneck 1984) was deployed 1 km east of the FVP disposal site. This instrumentation array measured currents (Marsh-McBirney two-axis electromagnetic current meter), temperature, salinity, and turbidity at one meter above the bottom, recorded in bursts at intervals of 15 min. Currents measured by the array are assumed representative of near-bottom currents over the entire study area, i.e., currents are spatially coherent. This is a good representation because of the relatively uniform topography in the area.

Dumping logs maintained by inspectors on disposal vessels were obtained from the New England Division, COE. These logs contain information on time, volume, and location of disposal. These records provided the timing, location, and initial mass of disposal plumes.

Estimates of disposal plume parameters were derived from previous studies (Gordon 1974; Custar and Wakeman 1977; Csanady 1981; and DeLoch and Waring 1984). We chose 50 m for initial plume width (standard deviation) and 0.3 cm/sec for the diffusion velocity and assumed that 0.25% of the disposed material remained in suspension.

RESULTS

Simulations were conducted with current meter data from three *in situ* instrumentation deployments that bracketed the disposal operations at the FVP site (26 April-14 May), and corresponded to periods of high disposal activity and dynamic changes in water column chemical exposures (Paul *et al.* 1987).

Behavior of the model can best be visualized by examining results of a single disposal event. Figure 2 shows the predicted trajectory of a plume created at the FVP site after a disposal vessel release on 30 April 1983 at 17:20. This disposal occurred shortly after high tide, with the initial plume transported to the east on the ebb tide. The plume was transported back and forth through the study area on subsequent flood and ebb tides while gradually drifting northeast.

The predicted temporal pattern of suspended solids at the FVP disposal location resulting from all disposal operations during the period 26 April-23 May is shown in Figure 3. Fifty-nine strong (>5 mg/l) peaks were predicted

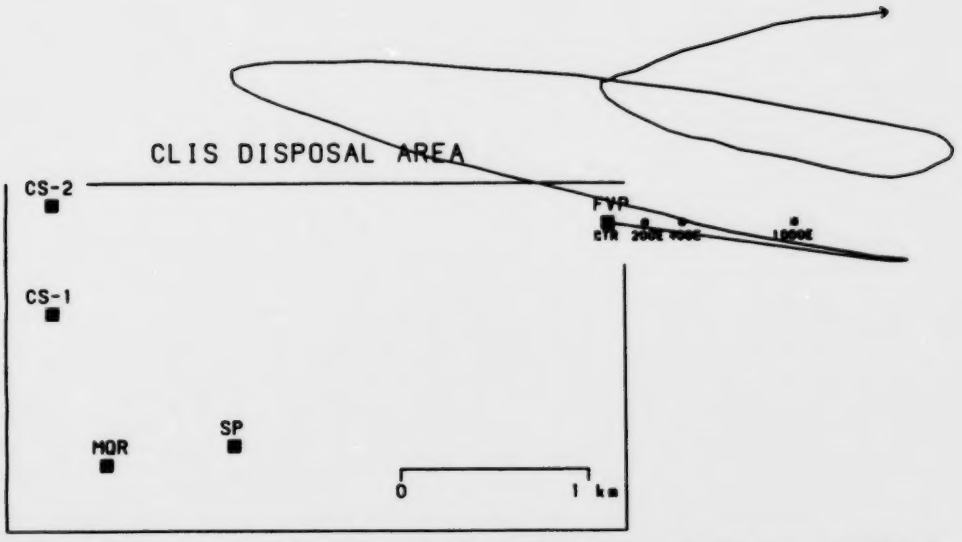

Figure 2. Predicted trajectory of disposal plume created at CTR on 30 April 1983 at 17:20. FVP study stations (CTR, 200E, 400E, 1000E) and active disposal sites (CS-1, CS-2, MQR, SP) are indicated.

as a result of the 149 dumping events.

Estimated time-averaged exposures due to disposal operations at all locations in the CLIS disposal site during the three in situ instrumentation deployments are shown in Table 1. The stations indicated in Table 1 correspond with

Table 1. Estimated time-averaged suspended solids concentration (mg/l) at FVP stations during 1983 DAISY deployment periods due to disposal operations in the Central Long Island Sound Disposal Site.

	CTR	400E	1000E	REFS
18-26 April	0.3	0.2	0.02	0.6
26 April - 23 May	0.8	0.7	0.4	0.0
26 April - 23 May disposal at CTR only	0.4	0.3	0.05	0.0
23 May - 16 June	1.1	1.2	1.6	0.02

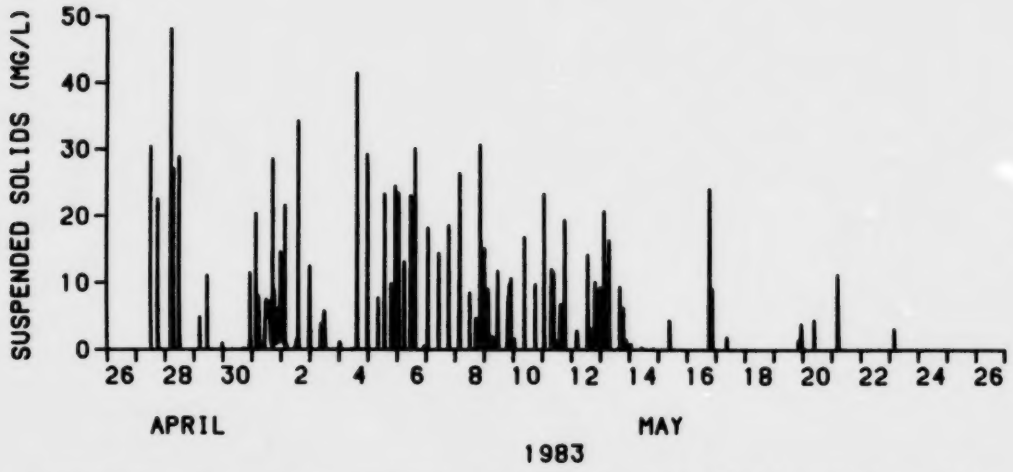

Figure 3. Temporal record at CTR of predicted suspended solids elevation above background resulting from disposal activities during 26 April-23 May 1983.

the FVP disposal site (CTR), stations 400 and 1000 m east of CTR (400E and 1000E), and the CLIS reference station (REFS) located 3 km southwest of CTR. Also shown in the table for the 26 April-23 May deployment are estimated exposures due only to disposal operations at the FVP disposal site (CTR).

DISCUSSION

Model predictions indicate that elevations of time-averaged suspended solids concentration at study stations reached 1.6 mg of dredged material per liter during spring of 1983. Predicted peak elevations approached 50 mg/l. Short term dispersion of plumes was primarily by the tidal component of the currents, while long-term dispersion depended upon residual current drift.

Simulations indicated that disposal operations unassociated with the FVP study influenced the exposure at FVP biological effects stations. The relative contribution

of these distant disposal operations was potentially large, as indicated by model results when disposal vessel operations were restricted solely to the FVP site (Table 1). The initial assumption of the FVP that the disposal mound would be the sole source of contamination is therefore suspect.

Time-averaged estimations of exposures to dredged material have been made independently using tissue residues of transplanted blue mussel, Mytilus edulis, deployed for one-month periods at FVP stations (Nelson et al. 1987). Mussel-derived exposures ranged between 0.0 and 0.9 mg/l, which are comparable to predicted elevations (Table 1).

Studies are underway to compare results obtained from this model with those from a model commonly used to predict dredged material dispersal, namely the Koh-Chang model (Brandsma and Divoky 1976). This comparison will serve as a benchmark, as well as a test for some of the underlying model assumptions.

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NEAR-SHORE VERSUS DEEP-SEA DISPOSAL OF WASTES AND DREDGED MATERIALS CONTAMINATED WITH PATHOGENIC BACTERIA

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ABSTRACT

Data collected over the past five years clearly demonstrate that a wide variety of gram-negative, heterotrophic bacteria, including human pathogens, remain viable but become non-culturable following their introduction into seawater. This phenomenon is generally accompanied by cellular miniaturization, loss of flagella, assumption of a coccoid shape, and, in certain cases, cell fragmentation and increased metabolic activity early in the transition. In contained systems, e.g., laboratory microcosms and in situ membrane chambers placed in sediment or in the water column, there is little or no decrease in total cell number. Viable bacteria, i.e., metabolically responsive cells, typically constitute 0.1 to 0.01% of the total cell number for a 10-day exposure to sea water. Although these bacteria are non-culturable on or in standard laboratory media, they retain ability to initiate disease in animals. This circumstance therefore contraindicates near-shore disposal of wastes and dredged materials contaminated with human pathogens. Such materials, and the water column and sediment to which they are added, will not show evidence of contamination, when examined with standard bacteriological tests for indicator bacteria and for pathogenic bacteria. Alternate tests, used in the laboratory demonstration of the viable but non-culturable phenomenon, should therefore be conducted on wastes and dredged materials, prior to disposal. These tests are rapid, specific microscopic methods that do not rely on growth of the bacteria being tested. Accumulation of viable, but non-culturable pathogens in near-shore habitats and subsequent introduction of those pathogens to pathways leading to human hosts, must be prevented and deep-sea disposal of pathogen-containing materials should be considered the preferable alternative. However, deep-sea disposal must be regarded as a short-term alternative, until such time as efficacious methods for disinfection can be developed.

BACKGROUND

The hypothesis that gram-negative, heterotrophic bacteria become non-culturable, i.e., change to a state in which they no longer exhibit visible growth in and on standard growth or culture media, and yet retain viability, was first proposed by Rita Colwell and a group of visiting scientists in her University of Maryland laboratory, in 1981. Work commenced with two representative bacteria, both frequently implicated in water-borne disease, Vibrio cholerae and Escherichia coli. Subsequent experiments did, indeed, demonstrate the existence of the viable but non-culturable state for V. cholerae and E. coli, and the data were first published by Xu et al. (1982). Since that time, with the notable exception of Pseudomonas sp. (Xu and Colwell, unpubl. data), the viable but non-culturable state has been demonstrated for an eclectic group of gram-negative, heterotrophic bacteria (Table 1).

Table 1. Genera known to exhibit a viable but non-culturable state when placed in non-nutritive media.

Genus	Citation
Aeromonas	Allen-Austin et al. (1984)
Agrobacterium	Xu and Colwell (unpubl.)
Campylobacter	Rollins and Colwell (1986)
Enterobacter	Xu and Colwell (unpubl.)
Escherichia	Grimes and Colwell (1986)
Klebsiella	Xu and Colwell (unpubl.)
Legionella	Hussong et al. (1987)
Salmonella	Roszak et al. (1984)
Shigella	Colwell et al. (1985)
Vibrio	Xu et al. (1982)

Clearly, this list is not all-inclusive but it suggests that the phenomenon may be a rather common theme amongst gram-negative, heterotrophic bacteria.

DEFINITION OF NON-CULTURABLE BACTERIA

The most frequently studied situation, with regard to viable but non-culturable bacteria, involves placing log or stationary phase, washed cells into sterile seawater. Morphological changes ensue, the most visible of which involve cells losing their flagella, rod-shaped cells assuming a coccoid shape, and all cells becoming reduced in size. These morphological events were early noted by several marine microbiologists (Morita, 1985), and were a fairly constant feature of the studies cited in Table 1. In some cases, cell fragmentation has been a prelude to reduction in size (Novitsky and Morita, 1978; Amy and Morita, 1983). Interestingly, some of the "reproductive organs of the bacteria" that were described by Lohnis and Fred (1923) are reminescent of miniature viable but non-culturable bacteria.

Changes in cellular morphology are accompanied by equally dramatic changes in cellular biochemistry and metabolic activity. Kjelleberg et al. (1983) reported an increase in metabolic activity that occurs early upon entry of Vibrio DW1 cultures into a starvation state. Novitsky and Morita (1977) observed that as the starvation/survival process continues, endogenous metabolism decreases

significantly. Initially at least, data reported for the starvation/survival and viable but non-culturable phenomena are compatible, and the two states are, in all probability, one in the same. As cells enter this state, membrane lipids undergo a cis- to trans- isomerization (Guckert et al., 1986). Recently, two laboratories have reported on changes that appear in cell envelope protein profiles, as cells become dormant, a word that for purposes of this paper will be used interchangeably with the terms starvation/survival and viable but non-culturable. Jaan et al. (1986) identified 2 new proteins that appeared in three marine isolates as they entered the starvation state; at the same time, 17 to 20 proteins disappeared. Groat and Matin (1986) also analyzed protein changes in starving cells, resolving at least 30 new polypeptides in the first 3 to 4 hours of starvation for E. coli. More recently, Schultz and Matin (1987) examined starvation protein synthesis in E. coli. From these data, one can hypothesize that the viable but non-culturable phenomenon is a regulated event. Superficially, at least, it can be argued that entry into a viable but non-culturable, vis a vis, dormant or starvation/survival, state is a gram-negative, heterotrophic bacteria survival mechanism analogous to sporulation in certain gram-positive bacteria. Recently, Roszak and Colwell (1987) reviewed survival strategies of bacteria, and suggested the term "somnicell" be used to describe the miniaturized, dormant stage of non-spore-forming bacteria.

Cultural changes that accompany entry of all species (Table 1) into the viable but non-culturable state are graphically summarized in Figure 1.

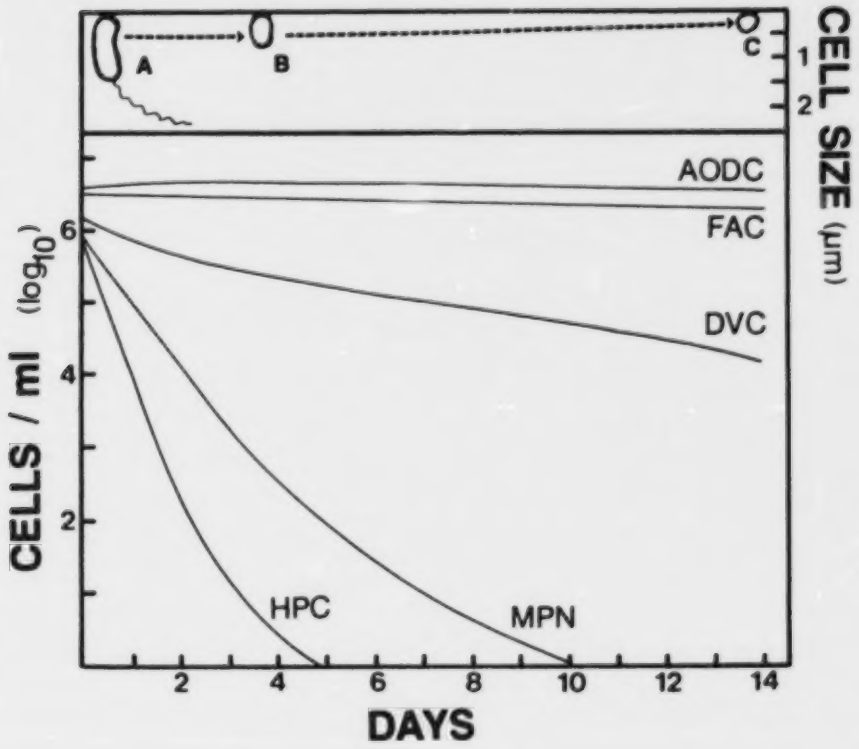

Figure 1. Representative survival curve of bacteria measured by acridine orange direct count (AODC), fluorescent antibody count (FAC), direct viable count (DVC), mean probable number (MPN), and heterotrophic plate count (HPC).

Total numbers of bacteria do not significantly change, whether measured by acridine orange direct count (AODC) or by fluorescent antibody count (FAC). As first pointed out by Xu et al. (1982), this supports the argument that cells do not loose cellular integrity. The direct viable count

(DVC), developed by Kogure et al. (1979), has been used as the test for physiological responsiveness amongst the species examined (Table 1), i.e., as a measure of viability. A flow diagram for the DVC procedure is presented in Figure 2, and details of the technique can be found in Kogure

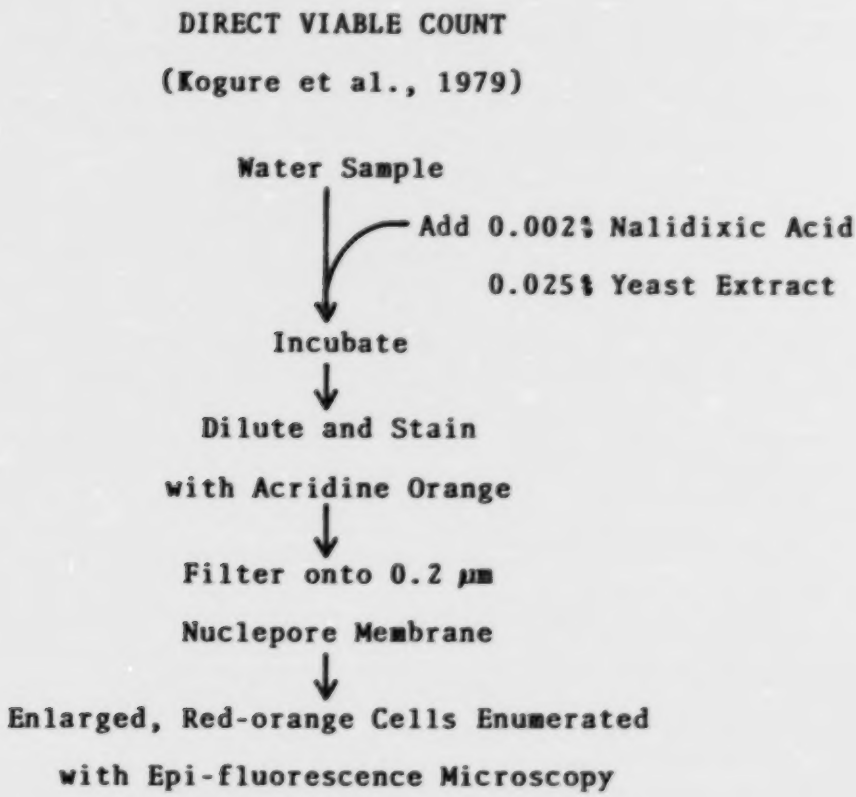

Figure 2. Direct viable count (DVC) protocol for detection of viable bacteria in water samples.

et al. (1979). Generally, there is a one to two-log reduction in the number of cells that retain viability as measured by the DVC. Another line of evidence to support the argument that the DVC is a true measure of viability was recently provided by Roszak (1985). In addition to exposing

cells to the DVC procedure, ^{3}H-thymidine was added to the system, and the presence or absence of label uptake determined by autoradiography. Clearly, enlarged cells, i.e., DVC-positive cells, had taken up the ^{3}H-thymidine label, as evidenced by the presence of black granules at the cell periphery (Roszak, 1985). As a further test of viability, ^{14}C-glutamic acid was substituted for the tritiated label, with similar results (Roszak, 1985). Finally, DVC-enlarged cells have also been observed to take up and reduce INT (Hernandez, unpubl.), a process that is indicative of respiration (Zimmerman et al., 1978).

While cells remain viable as measured by the DVC and by uptake of labelled substrate, they loose their ability to grow, and hence produce visible turbidity (broth) or colonies (agar), in and on all culture media normally supportive of growth of the species being tested. In only one case (Roszak et al., 1984), were the investigators able to resuscitate cells that had become non-culturable in estuarine (Potomac River) microcosms, and then only very early (at four days) in the non-culturable state. Generally, once cells enter the non-culturable state, they cannot be recovered with media conducive to normal, vegetative growth. Instead, recovery has depended on the use of animal models.

Animals that have proven successful in the recovery of viable but non-culturable bacteria include rabbits (Grimes and Colwell, 1986), mice (Brayton et al., 1984), embryonated

chicken eggs (Hussong et al., 1987), chickens (Pearson et al., unpubl.), and, most recently, human volunteers (Brayton et al., unpubl.). In every case, symptoms compatible with the clinical disease entity produced by the species being studied have been observed. Further, cells that were non-culturable upon introduction into the animal model were fully culturable and identifiable as the test strain when isolated from the model. Clearly, factors as yet unidentified are present in the animal models, including humans, that support return of the non-culturable cells to a culturable state.

Salient features of the viable but non-culturable state, discussed above, are summarized in Table 2. Further details about specific studies were recently reviewed by Grimes et al. (1986) and by Roszak and Colwell (1987).

Table 2. Characteristics of viable but non-culturable bacteria

Level	Characteristics
Culture	1. No growth on or in standard media 2. Resumption of growth under appropriate conditions
Cytology	1. Cells appear intact when stained with fluorochromes 2. Cells may undergo changes in size and/or shape 3. Cells may undergo reduction division
Biochemistry	1. Protein composition changes 2. Membrane lipids undergo change
Physiology	1. Endogenous metabolism decreases 2. Cells are substrate responsive
Virulence	1. Pathogens retain ability to cause disease 2. Agents of disease become culturable

NON-CULTURABLE BACTERIA IN SEDIMENT

The existence of viable but non-culturable bacteria in sediment has been hypothesized (Grimes et al., 1986) but not investigated, primarily because of the difficulty in identifying test strains in sediment. Therefore, in the summer of 1986, a field verification study was initiated to examine the fate of Vibrio cholerae ATCC 14035 (a cholera toxin-producing, 01 strain designated as the type strain) and Escherichia coli H10407 (an enterotoxigenic strain) in sediment from two sites in Chesapeake Bay. Nylon membrane (0.2 um, type ABA, Nuclepore) envelopes were constructed, filled with non-sterile sediment mixed with pure cultures of either V. cholerae 14035 or E. coli H10407, placed into a weighted open container, and lowered into position in the sediment of Chesapeake Bay at Solomons, Maryland (a clean site), and at Wingate, Maryland (a site near a crab processing plant, having sediment heavily enriched in chitin waste). Parallel studies were conducted in the laboratory using non-sterile sediment from each site that was inoculated with the two study strains and contained in two-liter polycarbonate graduate cylinders. Results of the field verification study were similar to those observed for water studies, with both strains becoming non-culturable between one and two months (Byrd, Grimes and Colwell, unpubl.). Generally, it was observed that V. cholerae entered the non-culturable state more rapidly in laboratory columns (one month) than in situ (two months). Unexpectedly, the presence of chitin in the sediment caused both test

strains to enter the viable but non-culturable state more rapidly. Previous work by Kaneko and Colwell (1975) had hypothesized that chitin prolonged survival of V. parahaemolyticus, an observation recently reported by Karunasagar et al. (1986). However, Karunasagar et al. (1986) only observed the survival of V. parahaemolyticus in flasks containing phosphate-buffered saline for 6 h, an amount of time insufficient to examine survival of an estuarine microorganism. Therefore, the precise effect of chitin on microbial survival in sediment cannot be determined at this time.

DREDGING AND DREDGED MATERIALS

The presence of viable but non-culturable bacteria in sediment has serious ramifications, with regard to dredging and disposal of dredged materials. It has been shown that both hydraulic dredging (Grimes, 1980), and clamshell dredging (Grimes, 1982), and the spoil deposition associated with each, resuspend both fecal indicator bacteria and specific enteric pathogens in numbers significantly greater than corresponding background levels in the water column. Although neither of these dredge studies tested for the resuspension of viable but non-culturable enteric pathogens, based on the Chesapeake Bay sediment study (Byrd, Grimes and Colwell, unpubl.), it can be hypothesized that such bacteria did exist in concentrations greater than those being detected by the conventional, cultural methods. Therefore, when dredging sediment with potential for contamination by

human pathogenic bacteria, both conventional, cultural tests and specific tests must be employed to ascertain the contamination level of the sediment. Specific tests could include FACs, fluorescent antibody-DVCs, or the newly-emerging gene probes. These tests are rapid, strain to species specific, and do not rely on growth of the organisms being tested for. Detection of pathogens, culturable or non-culturable, should then set into action certain precautionary measures, such as those outlined by Grimes (1980), as well as other actions, including capping, temporary closure of area shellfish beds, and temporary closure of the area to recreational use.

NEAR-SHORE VERSUS DEEP-SEA DISPOSAL

The preferable disposal alternative, but recognizably the most expensive alternative, is deep-sea disposal of wastes and dredged materials that have been shown to contain pathogenic microorganisms. Near-shore disposal of such wastes is not desirable, because it places pathogen-containing wastes in areas where humans can contact the pathogens and possibly develop disease. Pathways to humans could include either direct infection from water-borne microorganisms, e.g., as from swimming, diving, wind surfing, or by indirect contact, e.g., consumption of contaminated and improperly prepared seafood. The only possible way to contain wastes in the near-shore disposal operation would be to cap contaminated dredge spoil with

clean, non-contaminated material. This practice has had some success with regard to stabilizing spoil (Amson, 1987). However, Goyal and Adams (1984) demonstrated that pathogens survived for a considerable period of time (30 months) in sewage sludge at the Philadelphia dump site. The effect that capping would have on survival is unclear, although it can be argued that capping might render the pathogens inaccessible. Clearly, because dredging is a necessity, safe procedures, including disposal procedures, must be identified.

CONCLUSION

It is now clear that bacteria can enter a viable but non-culturable stage, but conditions for emergence from that stage are less than clear. While in this stage, they cannot be grown and as a consequence, detected by classic indirect enumeration methods. When introduced into a suitable animal host, viable but non-culturable bacteria can resume growth and, in the case of those that are pathogenic, initiate host colonization and disease. The implications of this phenomenon for ocean disposal of pathogen-containing wastes are clear. Such wastes must first be identified as to the type and magnitude of contamination, using rapid, direct detection methods such as FA-DVC. Once this has been done, wastes known to be contaminated can be handled and disposed of properly. If that disposal necessitates ocean disposal, then the wastes must be placed such that there is little or no opportunity for access to human pathways. This could

include deep-sea disposal, capping, disinfection prior to disposal, and incineration, depending on the waste type. It must not include disposal in estuaries and coastal zones where access to humans is almost certain.

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DECISIONMAKING FRAMEWORK FOR DREDGED MATERIAL MANAGEMENT

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Bioassay and associated tests have been developed and are being refined to assist in the evaluation of the environmental impact of dredged material disposal alternatives. Bioassay tests using aquatic animals, wetland plants and animals, and terrestrial plants and animals have been used to give appropriate information to describe the potential for contaminant mobility from dredged material into the biological components of disposal environments. Rainfall simulation-surface runoff water quality tests are being developed to determine contaminant mobility from dredged material placed in a terrestrial disposal environment. The potential for contaminant mobility resulting from physicochemical changes in dredged material following upland disposal has been determined. A management strategy and a decisionmaking framework have been developed that incorporate the above test results for selection of environmentally acceptable dredged material disposal alternatives.

MANUSCRIPT NOT AVAILABLE AT TIME OF PUBLICATION

WASTES IN MARINE ENVIRONMENTS

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Estuaries and coastal waters have borne the brunt of marine disposal activities, and many are in trouble around the United States. In contrast, the open ocean exhibits few adverse impacts from waste disposal, although it is unclear whether toxic pollutants will cause long-term damages to open-ocean resources.

Among disposal activities, industrial and municipal pipeline discharges are at least as important as dumping in causing damages in estuaries and coastal waters. Urban and agricultural runoff, although not classed as disposal activities, also are very important.

Discharges and dumping are regulated by the Clean Water Act and the Marine Protection, Research, and Sanctuaries Act. Programs and procedures established by these acts have helped reduce the amounts of pollutants entering marine waters. As implemented, however, these programs have not protected some estuaries and coastal waters from degradation. Moreover, even with total compliance, which is unlikely, existing regulations will not be enough -- many estuaries and some coastal waters will continue to degrade or begin to do so during the next few decades unless additional protective measures are taken.

Current efforts to identify those waters needing additional management are not systematic. The missing link is a framework for deciding where and how to provide additional management, either to reverse current degradation or to prevent degradation from occurring. Developing a systematic framework -- one using a broad, water quality-based approach to provide additional management where needed -- now seems appropriate.

Introduction

Every community and industry faces the question of where to dispose of waste materials. For years, marine environments have provided an answer for coastal communities and industries in the United States. In April 1987, the Office of Technology Assessment (OTA) released a comprehensive report on "Wastes in Marine Environments." This paper is based on the major conclusions of that report (and references therein), with a particular focus on estuaries and coastal waters.[1]

Estuaries and coastal waters are among the most important of all aquatic environments with respect to their commercial resources, recreational uses, and ecological roles. In 1985, for example, total commercial landings of fish and shellfish from all U.S. marine waters had a dockside value of $2.3 billion and a retail value several times greater. About one-half of this total was generated by fish and shellfish harvested within 3 miles of shore; in addition, many fish harvested from waters beyond the 3-mile limit depend on estuaries and coastal waters during portions of their life cycles.

Threats to Estuaries and Coastal Waters

Estuaries and coastal waters also are repositories for large amounts of pollutants derived from the disposal of various wastes and from runoff. In comparison with open ocean waters, estuaries and coastal waters bear the brunt of marine disposal activities. Almost 2000

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[1] OTA is a non-partisan, analytical support arm of the United States Congress. Copies of the report are available from the Superintendent of Documents, U.S. Government Printing Office, Washington, D.C. 20402-9325; GPO stock number 052-003-01062-3.

industrial and municipal pipelines discharge into estuaries and coastal waters. Thousands more discharge into rivers that flow to these waters or into municipal sewage treatment plants that later discharge into them. Large amounts of dredged material are dumped at numerous sites, and smaller amounts of sewage sludge and industrial acid wastes are dumped at a few sites. Runoff from agricultural lands and urban sites contributes significant quantities of pollutants in many areas.

As a result of these pollutant inputs, many estuaries and coastal waters around the country have suffered significant effects, and their overall health is declining or threatened. While some areas that once exhibited severe effects have improved greatly in recent years, noticeable deterioration continues to occur or is accelerating in others. Attention has focused on well-documented problems in the northeastern United States (including the Chesapeake Bay and the New York Bight), southern California, and Puget Sound. Serious impacts, however, also have occurred in the less-studied Gulf of Mexico and the southeastern United States. In contrast, the open ocean exhibits few adverse impacts from waste disposal, although it is unclear whether toxic pollutants will cause long-term damages to open-ocean resources.

The extent of degradation varies greatly around the country -- in type, spatial scale, duration, and commercial importance -- and includes:

- changes in water quality (eutrophication, hypoxia, turbidity, elevated concentrations of pollutants);
- loss of submerged aquatic vegetation;
- impacts on fish and shellfish (bioaccumulation of chemicals, disease and abnormalities, reproductive failure, mortality);
- impacts on entire marine communities (changes in diversity and abundance);
- closures of beaches and shellfish grounds because of microbial or chemical contamination;
- a rising incidence of reported human disease, from consumption of contaminated shellfish or swimming in contaminated marine waters; and
- accumulation of toxic pollutants in sediments.

Among disposal activities, OTA concluded that industrial and municipal pipeline discharges are at least as important as dumping in causing damages in estuaries and coastal waters. Urban and agricultural runoff, although not classed as disposal activities, also are very important (in some cases more important than discharges). Furthermore, other activities not discussed in detail in the OTA report -- such as freshwater diversion, coastal development, habitat destruction, and overharvesting -- can be significant factors in the degradation of our estuaries and coastal waters.

To regulate discharges and dumping, the Federal government enacted the Clean Water Act and the Marine Protection, Research, and Sanctuaries Act (MPRSA), respectively. The pollutant control programs and permitting procedures developed under these acts have helped achieve significant reductions in the amounts of some pollutants (for example, suspended solids, nutrients, and oxygen-demanding substances) entering many marine waters.

As implemented, however, these programs have not protected some estuaries and coastal waters from degradation. Moreover, even with total compliance, which is unlikely, existing regulations will not be enough. For example, current regulations do not adequately address toxic metals and organic chemicals and they barely touch on the ubiquitous problems associated with runoff.

In addition, pressures to continue current disposal activities in estuaries and coastal waters will increase because of: 1) increased coastal population and development; 2) the frequently lower costs of marine disposal; and 3) limits on the availability of land-based disposal options due to increased public opposition, regulatory restrictions, and

economic constraints. Even with extensive waste reduction and reuse efforts, large amounts of municipal wastes, industrial wastes, and dredged material will continue to require disposal for the foreseeable future.

As a result, many estuaries and some coastal waters (including some that exhibited improvements in the past) will continue to degrade or begin to do so during the next few decades unless additional protective measures are taken.

Protecting Estuaries and Coastal Waters in the Future

If we desire to maintain or improve the health of estuaries and coastal waters, it will be essential to increase efforts to reduce the levels of pollutants in municipal and industrial discharges, to reduce runoff where necessary, and to minimize waste disposal in estuaries and coastal waters wherever possible.

The ability to minimize disposal in these waters, however, may be precluded by policy decisions or technical factors related to disposal on land or in the open ocean. For example, one option to minimize inputs is to shift some disposal activities to the open ocean. However, most pipelines cannot be extended much further offshore and out of estuaries and coastal waters, especially on the east and Gulf coasts. Furthermore, shifting pipeline discharges (where feasible) and dumping to the open ocean would run counter to current attitudes about open ocean disposal. These factors and policies are discussed in more detail in the OTA report.

Thus, some other options for minimizing waste disposal and pollutant inputs in estuaries and coastal waters deserve more attention. First, continued and enhanced implementation of the present pollutant control

programs and permitting procedures could be pursued. This approach will likely be sufficient in many areas, but it will require some combination of continued Federal, State, and local investments in the construction of municipal treatment plants, enforcement efforts, and monitoring and research. In addition, pollutant controls could be expanded to cover a broader range of toxic pollutants and industrial sources.

Even these improvements in current pollutant control programs will not be sufficient to protect all estuaries and coastal waters, as discussed above. Second, then, is the need to supplement current programs by developing additional means of waterbody management in many areas. The Federal government and some States have recognized this need and have developed "waterbody management" programs for some estuaries and coastal waters -- the Chesapeake Bay Program and the Puget Sound Water Quality Authority being noteworthy examples.

In general, individual waterbody management programs attempt to bring together all appropriate parties (including ones that have authority to deal with runoff and land-use issues), identify the most important problems and their causes, and devise a management plan that identifies site-specific goals and discusses how resources could be used effectively to achieve those goals.

These waterbody management efforts appear promising, but they exist for only a few estuaries and coastal waters. Mechanisms to spur the development and adequate design of such programs are in need of further attention. One fundamental and critical link is missing -- a systematic framework for deciding which estuaries and coastal waters need help, either to reverse current degradation or to prevent significant degradation from occurring.

Developing a framework -- whether at the Federal level or at the individual State level -- to identify waterbodies and help initiate management plans now seems appropriate. It could be based on an expanded version of the water quality approach that currently is used more extensively for freshwater systems but only to a limited extent for marine waters.

Some components or models for such a framework already exist in various institutional guises. At the Federal level, for example, these include: 1) EPA's National Estuary Program (which was formally authorized by the Water Quality Act of 1987) and EPA's internal initiative on Near-Coastal Waters; 2) Federal participation in the Chesapeake Bay Program; 3) the Coastal Zone Management Act; and 4) various planning areawide and watershed planning mechanisms in the Clean Water Act. Appropriate models also have been developed by various coastal States, as well.

The major questions are whether these and other mechanisms at all governmental levels will be used effectively and whether enough waterbody management programs will be implemented. The answer will depend to a great extent on our ability to achieve cooperation among responsible agencies, involve the public in the decision-making process, provide sufficient funding, and make difficult decisions about land-use issues and coastal development practices.

More than anything else, implementing a systematic framework may require consolidation of these sometimes disparate efforts and possibly some explicit Federal oversight or direction in the development of other components. Without such efforts, however, many estuaries and coastal waters are likely to suffer more degradation.

The Role Of Sediment Quality Values In Management Decisionmaking

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ABSTRACT

Although no standard approach for setting and interpreting sediment chemical levels is available at the national level, an effects-based approach for using sediment chemistry data in management decisionmaking is currently being evaluated in Puget Sound. The Apparent Effects Threshold (AET) approach relies on field and laboratory effects data to determine chemical-specific sediment quality values. By definition, the AET is the chemical concentration in sediment above which statistically significant adverse effects are always expected in a biological indicator. AET can be developed for any kind of biological response, including laboratory and field indicators. At present, AET have been developed using three bioassays, the abundance of benthic infauna, and bioaccumulation.

Several uses of sediment quality values generated by or modified from the AET approach are being evaluated. These include ranking of sediments to identify problem areas, setting remedial action cleanup goals for contaminated sediments, and initial identification of dredged sediments that may be acceptable for unconfined, open-water disposal. Ranking of sediments within Puget Sound is based on comparison of chemical concentrations in sediments from study areas to the AET. Cleanup goals for marine sediments are based on comparison of sediment chemical concentrations to AET of specific biological indicators. Initial testing of the technical acceptability of dredged sediment for unconfined, open-water disposal is based on a comparison of sediment chemical concentrations to "screening level" and "maximum level" values. Maximum level values are based on the highest AET for a range of biological indicators while screening level values are based on sediment chemical concentrations approximating "clean" reference areas within Puget Sound. Dredged sediment having chemical concentrations at or below the screening level values is considered technically suitable for unconfined, open-water disposal. Dredged sediment having chemical concentrations above the maximum level values is not considered suitable for unconfined, open-water disposal. Dredged sediment having chemical concentrations between

the screening and maximum level must be biologically tested before a decision on suitability for open-water disposal can be made.

INTRODUCTION

Sediments are an extremely important reservoir for chemical contaminants discharged into marine and estuarine environments. The management of contaminated sediments requires a decisionmaking framework and associated criteria to evaluate the relative impact to the environment or to public health. Such criteria may be based on measured biological tests (eg. bioassays) of individual sediment samples or on the predicted effects of the measured chemical contaminants in the sediments. Each of these distinctive approaches to developing sediment quality criteria has advantages and disadvantages. The former approach provides a direct assessment of the biological hazard of each sediment sample. However, this approach requires that each sample be subjected to one or more designated biological tests. The latter approach involves an indirect assessment of biological impact based only on the chemical charactistics of the sediments. Its advantage is that specific numerical criteria can be developed that only require information on the chemical characteristics of the sediments. This approach can therefore take advantage of historical chemical data and does not require biological assessment of each sediment sample.

The integration of sediment criteria into environmental decisionmaking processes requires consideration of a wide range of technical and policy issues. At the national level, there is debate on the role that chemistry data should play in evaluating the quality of marine and estuarine sediments. Some argue that past attempts to use chemistry data in marine programs (primarily dredging programs) have not yielded

insights into sediment quality and the role that any given sediment will play with its unique set of contaminants will play in toxicity to benthic organisms. Even among programs that apply chemistry data in the evaluation of sediment to be dredged, a variety of opinions exist on how these data should be interpreted in the decisionmaking process. At one end of the spectrum, chemistry data are required as general information only and are not used to base decisions on the technical suitability of dredged material for unconfined, open-water disposal. Still others require chemistry data only in exceptional circumstances and are used sparingly in the decisionmaking process. At the other end of the spectrum, some programs require data for a select group of chemicals which are then compared to strict pass/fail criteria in the decisionmaking process.

The purpose of this paper is to present an approach to setting sediment quality values and discuss several potential uses of sediment chemistry data in decisionmaking. Uses include ranking of sediments to identify problem areas, setting remedial action cleanup goals for contaminated sediments, and initial identification of dredged sediments that may be acceptable for unconfined, open-water disposal.

RELATIONSHIP BETWEEN SEDIMENT CHEMISTRY AND SEDIMENT TOXICITY

The lack of consistency in the application of sediment chemistry data in decisionmaking reflects the technical uncertainty associated with interpreting the relationship between chemical concentrations and toxicity. Historically, sediment chemistry data have been collected for a very few chemicals of concern (mostly metals) and these concentrations were evaluated

based on a comparison to concentrations of the same compounds from reference areas. When concentrations in the test sediment were found to be elevated above the reference values this was interpreted to indicate potential environmental risk. Often times, however, when the same sediment was tested using accepted bioassay protocols, no biological effects were noted. This result has led to uncertainty that sediment chemical concentrations can be reliably used to establish toxicity potential.

Much of this uncertainty arises because of the difficulty in developing a mechanistic (e.g., cause and effect) theory that provides a robust understanding of the relationship between sediment toxicity and sediment chemical concentrations. By in large, these theories center around the assumption that sediment toxicity is associated with the concentration of chemicals found in interstitial water, and must account for several mitigating factors (Battelle 1985). Theories currently being evaluated are equilibrium partitioning theory (using sediment-water interactions, Karickhoff et al. 1979; and using sediment-biota interactions, Bolton et al. 1985). Adding to this uncertainty is the lack of a comprehensive database containing complete sediment chemistry and biological testing results (Neff et al. 1986). Such a database is needed to verify model results for a range of conditions.

Finally, when sediment chemistry data have been used in decisionmaking (primarily in dredging programs), they have often been interpreted with a single "pass/fail" value given to each chemical. The use of such standards implies a more complete understanding of relationships is such that definitive relationships between complex chemical mixtures and biological interactions than is currently available.

In addressing the issue of sediment quality in Puget Sound, it has been recognized that determining what degree of contamination is "acceptable" is not a simple "black or white" decision. People hold very different views on what constitutes an "acceptable" effect on the environment. Uncertainties in scientific information about the relationship between chemistry and sediment toxicity further emphasizes this "gray" area that exists in interpreting what is "acceptable".

Sediment chemistry represents a continuum of sediment quality. As such there are points along the continuum at which contaminant concentrations are low enough that the sediment does not present an unacceptable adverse impact to the biota ("white"). Conversely, there is a point along the continuum at which a contaminant concentration is high enough that adverse impacts are assured ("black"). Between these two points on the continuum is a wide range of contaminant concentrations for which the impact on biota is not certain ("gray").

In its simplest form, the lower end of the continuum can be represented by sediments in which all contaminants are at concentrations that are below their detection limit, while the upper range can be represented by concentrations greater than those defined as hazardous waste. Although these points along the continuum represent conservative estimators of no impact and definite impact conditions respectively, they do not provide much use in the decisionmaking process. Virtually all sediment contains at least one chemical present at a concentration above feasible detection limits, and sediment classified as hazardous waste is already considered harmful to the environment. The task is to refine the lower and upper chemical concentrations that could serve to define sediment quality.

A variety of approaches were considered for setting sediment quality values for Puget Sound. The regulatory

agencies recognized a need to validate sediment quality values based on an extensive region-specific, quality-assured database containing chemical and biological effects information on a wide variety of sediments in Puget Sound. The original database was developed for the Commencement Bay Remedial Investigation (Barrick et al. 1985). To this database were added quality-assured data from other areas of Puget Sound (Beller et al. 1986).

Information contained in the database included chemistry analyses on 71 chemicals and the results of multiple species bioassays using the same sediments (Barrick et al. 1985). The bioassays included a 10-day amphipod test (Schwartz et al. 1985), an oyster larvae test (Chapman and Morgan 1983), and the Microtox test (Schiewe et al. 1985; Williams et al. 1986). Benthic infaunal abundance was also included as a direct measure of effects in sediment.

The sediment chemistry/biological effects data were analyzed in a variety of ways in an attempt to develop a technique that was efficient at predicting biological impacts based on sediment chemical concentrations. Among the approaches tested included two applying equilibrium partitioning theory, one applying water quality criteria to interstitial water (developed by EPA Region VI), one using a bioassay (Swartz et al. 1985), one that based sediment quality on the presence/absence of benthic species (Screening Level Criteria, Battelle 1985), and one which combined both bioassay data and measurements of benthic community health (Apparent Effects Threshold, Barrick et al. 1985).

Of the approaches considered, the Apparent Effects Threshold (AET) approach was found to be the most sensitive at predicting sediments in which biological effects were actually observed (Beller et al. 1985).

APPARENT EFFECTS THRESHOLD APPROACH

The AET is the chemical concentration in sediments above which statistically significant biological effects are always expected. The focus of the AET approach is to identify concentrations of contaminants that are associated exclusively with sediments having statistically significant adverse biological effects (relative to reference sediments). The approach can be used for any chemical and for any observable biological effects (e.g., bioassays, infaunal abundances at various taxonomic levels, bioaccumulation). A pictorial representation of the AET approach for two chemicals is presented in Figures 1 and 2 for a subset of the data for amphipod bioassay results in Puget Sound. Two subpopulations of all sediments analyzed for chemistry and biological effects are represented by bars in the figures, and include:

- Sediments that did not exhibit significant toxicity (e.g. to amphipods in bioassays)

- Sediments that exhibited statistically significant ($P<0.05$) toxicity in bioassays.

The horizontal axis in each figure represents sedimentary concentrations of contaminant of concern (i.e., lead or 4-methyl phenol) on a log scale. The AET for lead was based on lead concentration ranges corresponding to sediments that did not exhibit significant biological effects. The AET for 4-methyl phenol were determined in a similar manner.

AET correspond to concentrations above which <u>all</u> samples for a particular biological indicator were observed to have adverse effects. Data are treated in this manner to reduce the weight given to samples in

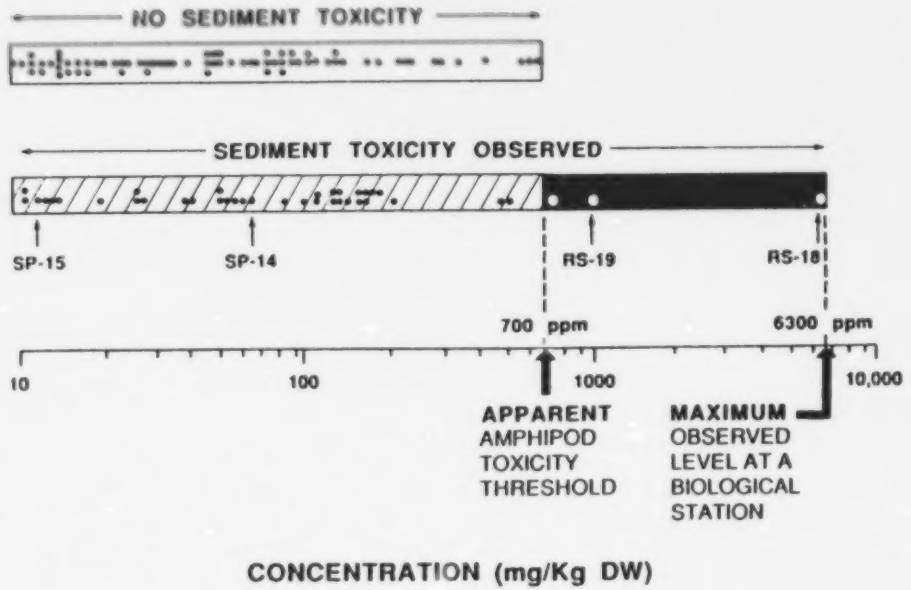

Figure 1. The AET approach applied to sediments tested for lead concentration and amphipod mortality during bioassays.

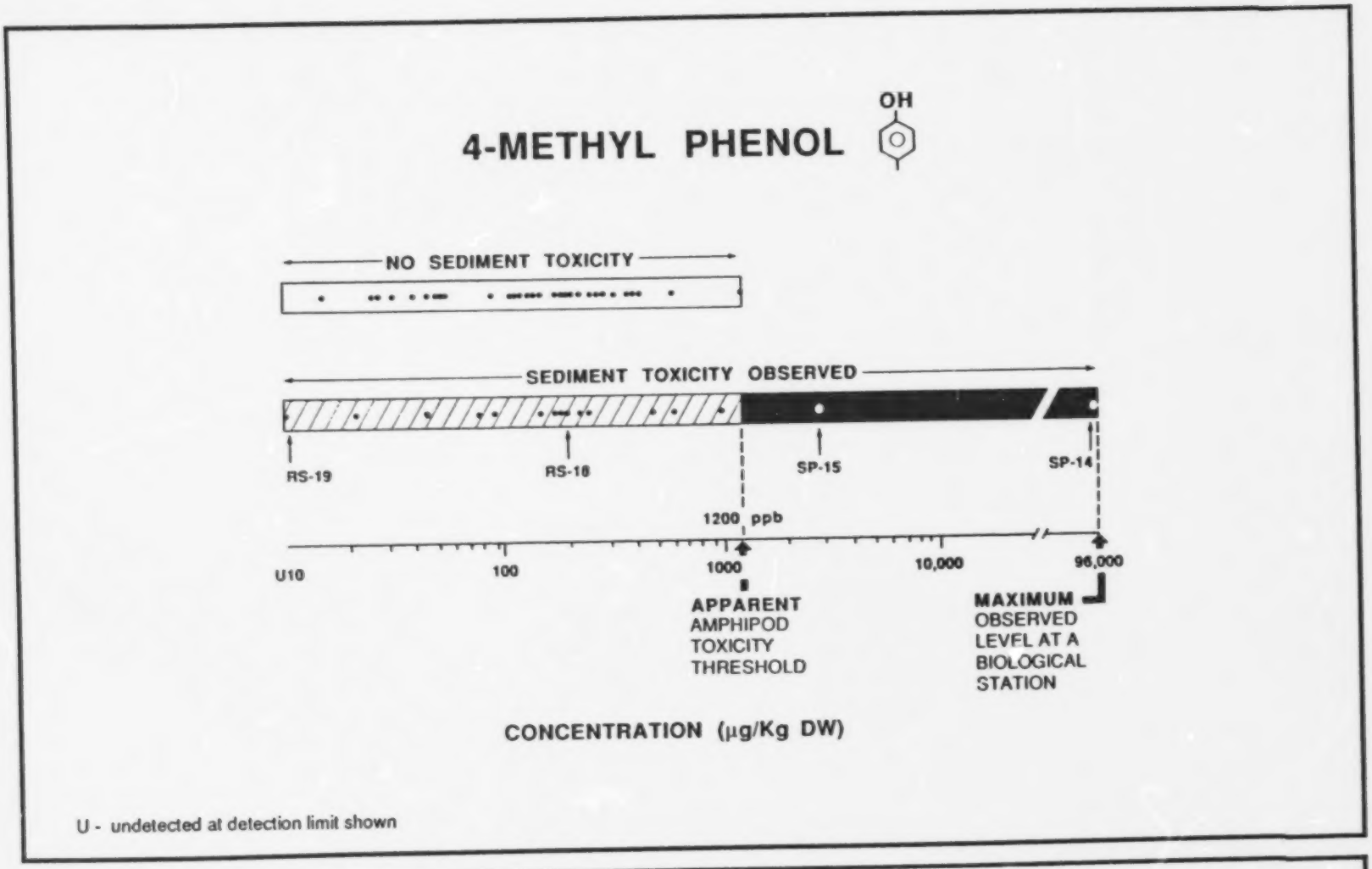

Figure 2. The AET approach applied to sediments tested for 4-methyl phenol concentration and amphipod mortality during bioassays.

which factors other than the contaminant examined (e.g., *other contaminants, environmental variables*) may be responsible for the biological effect. For example, sediment from Station SP-14 shown in Figure 2 exhibited severe toxicity, potentially related to a greatly elevated level of 4-methyl phenol (7,400 times reference levels). The same sediment from Station SP-14 contained a low concentration of lead that was not critical in establishing the AET for lead (Figure 1). Despite toxic effects displayed by the sample, sediments with higher lead concentrations exhibited no statistically significant biological effects. These results were interpreted to suggest that the effects at Station SP-14 were more likely associated with 4-methyl phenol (or a substance with a similar environmental distribution) than with lead. A converse argument can be made for lead and 4-methyl phenol in sediments from Station RS-18. Hence, the AET approach helps to identify different contaminants in a complex mixture of contaminants that are most likely associated with observed effects at each biologically impacted site.

If an unmeasured chemical (or group of chemicals) is not distributed in the environment in the same way as a measured chemical (e.g., if a certain industrial process releases an unusual mixture of contaminants), the effect should be discerned if a sufficiently large data set is used to establish AET. Because AET are set by the highest concentration of a given chemical in samples <u>without</u> observed biological impacts, AET will not be affected by less contaminated samples in which unmeasured chemicals cause biological impacts.

If an unmeasured toxic chemical is distributed differently in the environment from any of the measured chemicals, it is unlikely that the AET approach could predict impacts at stations where the chemical is indicating toxic effects. However, the predictive

success of AET can be tested using an independent field data set. Such a test determined that AET identified from 82 to 94 percent of the impacted stations, when the biological indicator was oyster larvae bioassays, Microtox bioassays, or depressions in benthic infaunal abundances (Beller et al. 1986). Lower success was obtained with the amphipod bioassay (54 percent of the impacted stations were identified), which may be related to an apparent sensitivity of the amphipod bioassay to some fine-grained sediments even in the absence of contamination (Beller et al. 1986; Ott 1985).

AET can be developed for any number of biological response indicators including laboratory and field indicators. As discussed earlier, four biological response indicators (amphipod bioassay, oyster larvae bioassay, Microtox bioassay, and benthic infaunal abundances) were used to develop AET for each chemical measured in the sediment. The various AET for each chemical can be very similar or can be very different, depending upon the relative sensitivity of each biological indicator to different sediment conditions. With most chemicals four different AET were developed, each representing a different biological indicator. For each chemical, the AET could be viewed as a ranking system, in which the lowest AET represents the concentration at which species could be impacted by the chemical, while the highest AET represents the concentration at which even relatively less chemically sensitive species would be impacted.

APPLICATION OF SEDIMENT QUALITY VALUES

To date, sediment quality values based on AET have been proposed for use in several Puget Sound programs. Programs include remedial investigations, feasibility studies, and dredged material evaluation.

Remedial Investigations--

AET were used to identify and rank toxicity of estuarine sediments in Commencement Bay (WA) which had been targeted in 1981 for action under Superfund. Field studies for Commencement Bay had been designed to document the degree and spatial extent of chemical contamination, adverse biological effects, and potential threats to public health. These studies resulted in a large and complex database with multiple levels of interpretation possible. Contamination had been measured in three media: bottom sediments, water column particles, and tissue of biota. Four kinds of biological effects potentially related to sediment contamination were evaluated: alteration of benthic macroinvertebrate assemblages, toxicity to bioassay organisms (amphipod, oyster larvae), prevalence of histopathological disorders in English sole livers, bioaccumulation (English sole and cancrid crab muscle tissue, English sole livers). AET were used to identify and rank problem areas within Commencement Bay and to set an order of priority for site cleanup.

Feasibility Studies--

AET have also been used to recommend sediment cleanup criteria for the implementation of remedial action alternatives in identified problem areas in Commencement Bay. The goal of sediment remedial action is to alleviate contamination in problem areas and thereby to eliminate associated adverse biological effects. Based on AET, exposure to sediments at target concentrations (following cleanup) is not expected to result in adverse biological effects.

Three levels of sediment cleanup criteria were recommended for use during the feasibility study. In the event that the primary criteria for certain chemicals were found to be infeasible, then less stringent

alternative criteria could be applied. If these criteria also proved to be infeasible, then a third, less stringent set of criteria could be applied. The three levels of sediment cleanup criteria are:

- Target Cleanup Criteria - these sediment contaminant concentrations are not predicted to result in adverse biological effects. Adverse effects are defined as the statistically significant effects (i.e., significant responses in amphipod mortality, oyster larvae abnormality, or Microtox bioassays; depressions in the abundances of higher taxonomic groups of benthic infauna; significant PCB bioaccumulation in fish). Target cleanup criteria are based on the lowest AET among four of the more sensitive biological indicators (e.g., Microtox, oyster larvae, abundance of benthic infauna, and PCB bioaccumulation)

- Alternative Level 1 Criteria - these concentrations are predicted to protect against certain site-specific biological effects, but not for as wide range of adverse biological effects as the target cleanup criteria. Specifically, alternative level 1 criteria are designed to protect against acute lethal toxicity and depressions of a relatively wide range of indigenous benthic infauna. Alternative level 1 criteria are based on the lower of two AET; Benthic infauna and amphipod bioassay)

- Alternative Level 2 Criteria - these are upper limits for allowable concentrations above which multiple biological effects or severe biological effects (defined previously) would be expected. Alternative 2 criteria are based on the higher of the amphipod or benthic AET or the lower of the "severe effects" AET. The "severe effects" AET is defined as >50 percent response in a bioassay (i.e., >50 percent mortality in amphipod bioassays) or the occurrence of significant benthic infaunal depressions in more than one major taxonomic group (i.e., two or more depressions among Mollusca, Crustacea, and Polychaeta).

Dredged Material Evaluation--

Sediment quality values for dredging were developed applying AET by the Puget Sound Dredged Disposal Analysis (PSDDA) study which is being conducted by the federal and state regulatory agencies involved in making decisions on dredged material disposal in Puget Sound. The purpose of the sediment quality values developed by PSDDA was to provide chemical concentrations at which sediments can be screened to determine whether biological testing is warranted in order to make a decision on the suitability of the sediment for unconfined, open water disposal. Both maximum level and screening level values were developed. PSDDA chose the maximum level (concentration of a chemical at which effects are assured) to be represented by the highest AET for a range of biological indicators. The highest AET provides a conservative value for the maximum level because it is, by definition, that concentration above

which adverse biological effects were always observed in all four biological indicators tested.

In setting the screening level, the AET were not as directly useful, since even the low AET represents a chemical concentration that has been related to biological effects for the most sensitive bioassay species. PSSDA relied on several sources of information to develop the screening level values for sediment quality. Initially the screening level was set at a chemical concentration that was 10 percent of the highest AET. These values were, in turn, examined on a chemical-by-chemical basis to ensure that they were below the low AET. The values were then compared to chemical concentrations of sediment from sites within Puget Sound that are accepted as areas having no biological impacts (reference sites). These sites were typically removed from historical or presently known sources of pollution. If the screening level value was found to be below the reference site values, the screening level value for that chemical was increased to the reference site value. The linking of reference values and the lowest AET provided a conservative baseline at which chemical concentrations were acceptable.

An important feature of the sediment quality values developed by PSDDA is the separation of dredging project "pass" (acceptable for unconfined disposal) and "fail" (unacceptable for unconfined disposal) values. Figure 3 shows the relation of the PSDDA values to reference site values and Puget Sound interim criteria for one chemical of concern. Dredged material evaluation in Puget Sound is currently based on interim criteria that were developed in response to local permit requirements of the cities of Seattle, WA (for use of Fourmile Rock disposal site; USEPA 1984) and Everett, WA (for use of the Port Gardner disposal site; USEPA 1985). These cities have shoreline jurisdiction over two Puget Sound high use

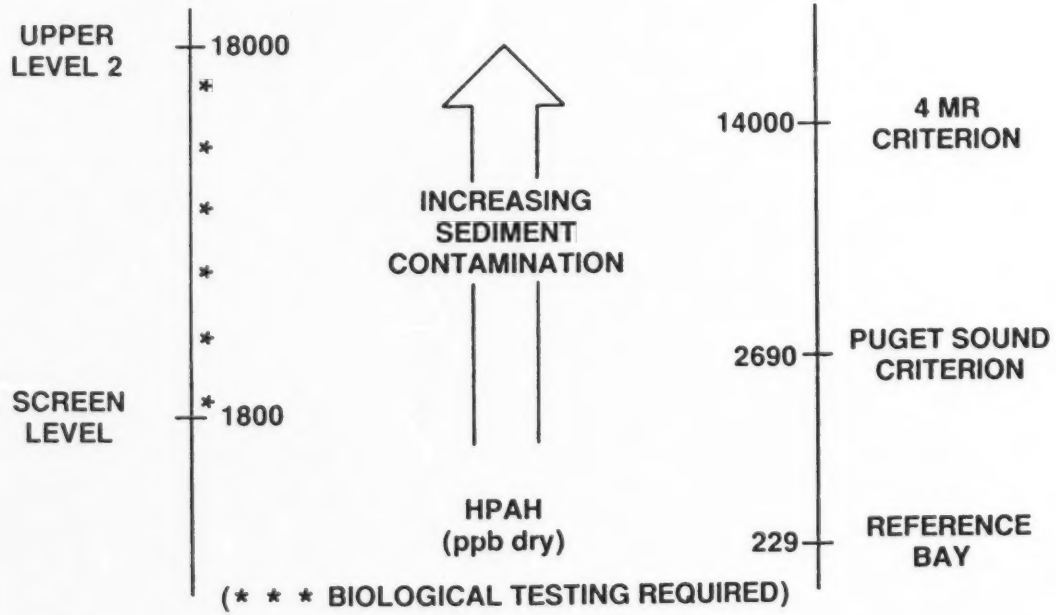

Figure 3. Comparison of PSDDA screening level and maximum level (e.g., upper level) for high molecular PAH to other sediment quality values in Puget Sound.

open-water disposal sites. The Fourmile Rock interim criteria include pass/fail sediment criteria that are based on chemical concentrations presently found at the disposal site. The Puget Sound interim criteria (which are to be applied to dredged material going to all other Puget Sound open-water disposal sites) include pass/fail sediment criteria that are based on chemical concentrations presently found in the central basin of the Sound. In most cases, the PSDDA screening level is below the two interim criteria that have been set for dredged material evaluation (e.g., Fourmile Rock Interim Criteria and Puget Sound Interim Criteria), while the maximum level is above the Fourmile Rock values. For example (Figure 3), high molecular weight polyaromatic hydrocarbons (HPAH) have a screening level value of 1800 ppb, a maximum level value of 18000 ppb. Under the interim criteria, a sediment containing concentrations of 2690 ppb (for Puget Sound criteria) or 14000 ppb (for Fourmile Rock criteria) would not be allowed for unconfined, open water disposal, while under PSDDA any sediment having concentrations between 1800 ppb and 18000 ppb would require biological testing. Typically, sediments having chemical concentrations above the maximum level would not be considered suitable for unconfined, open-water disposal based on chemical testing alone. However, if the dredging applicant feels that project sediments being tested are anomalous (e.g., maximum level chemical values are not predictive of biological effects in that sediment), then extensive biological testing can be conducted to determine suitability for unconfined, open-water disposal.

VERIFICATION OF AET-DERIVED SEDIMENT QUALITY VALUES

To date, the utility of AET in developing sediment quality values has been evaluated in a number of ways. For one, additional studies are being conducted within

Puget Sound under the Puget Sound Urban Bays Program (sponsored by EPA Region X) in which data on sediment chemistry and biological indicators are being collected. As these databases are completed and quality assured, they are added to the Puget Sound sediment quality database. Prior to their addition, this database is used to evaluate the predictive ability of AET. Among other evaluations, the database is used to determine how well the AET can predict impacted stations. In general, the predictive ability has been between 82 and 94 percent depending upon the biological indicator being evaluated (Beller et al. 1986).

The AET have also been applied to databases from other regions of the country to determine predictive accuracy. Two databases have been evaluated, one from San Francisco Bay (9 samples; Chapman et al. 1986, 1987), and one from southern California coast (71 samples; Word and Mearns 1979). In the San Francisco Bay study, laboratory sediment bioassays and benthic infaunal assemblages were used as biological indicators. One or more chemicals were found to exceed their AET at two of the nine stations sampled in San Francisco Bay. Biological effects were found at both of these stations. In this respect the AET was predictive of biological impacts. Biological effects were also found at three of the nine stations at which no chemical exceeded the Puget Sound derived AET. In two of these three stations only the mussel larvae bioassay showed impacts.

In the southern California study, biological indicators included an ifaunal index and echinoderm abundance. In all 71 stations were analyzed. One or more chemicals exceeded their AET at 13 of the 71 stations. All of these stations were located in the immediate vicinity of municipal sewage outfalls at Santa Monica, off Palos Verdes, in San Pedro Bay, and off Pt. Loma. Biological effects (reductions in infauna index or

echinoderm abundance) were found at all but 1 of the 13 stations at which an AET was exceeded.

Finally, the screening level and maximum level values proposed for use by PSDDA have been evaluated using data collected from dredged material evaluation tests conducted for four projects. In all, 61 sediment samples were taken in which chemical testing and at least one bioassay (amphipod bioassay) were conducted. For each sample, sediment chemistry data were compared to the screening and maximum level values. Each sample was identified as being either below the screening level in sediment chemistry, above the maximum level, or between the screening and maximum level. Using the amphipod bioassay data, the management utility of the screening level and maximum level guidelines was evaluated. In all sediment samples in which the sediment chemistry was above the maximum level (6 samples), significant amphipod mortality above controls was noted. In all sediment samples in which sediment chemistry was below the screening level (15 samples), no significant mortality compared to mortality in controls was found. In the remaining samples (41 samples), sediment chemistry was between the screening and maximum level guideline values. In these samples, significant mortality in the amphipod bioassays was noted in 9 samples, while no significant mortality was noted in the other 31 samples.

Although the results of the verification exercises evaluating the predictive ability are promising, more verification will be conducted to establish the utility of AET in establishing sediment quality values. In addition, more data sets from other regions of the country are needed in verification exercises to demonstrate the transferability of AET-generated sediment quality values from Puget Sound to other regions of the country.

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The U.S. EPA Ocean Data Evaluation System

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ABSTRACT

Section 301(h) of the Clean Water Act allows publicly owned treatment works to receive variances from secondary treatment requirements for discharges to marine waters when certain environmental conditions are met. The 301(h) program is administrated by the United States Environmental Protection Agency (U.S. EPA) through the issuance of 301(h) modified National Pollutant Discharge Elimination System (NPDES) permits, permit compliance determinations and permit reissuance. Included in the 301(h) modified NPDES permit are specific requirements for biological, water quality, and influent/effluent monitoring.

The Ocean Data Evaluation System (ODES) is a user-friendly computer system developed by the U.S. EPA for storing and analyzing marine environmental data collected by the permitted dischargers. ODES was developed to handle a variety of data types. ODES also provides a powerful set of statistical, graphical and modeling tools that are being used to determine permit compliance. Planned uses of ODES include the evaluation of the effectiveness of 301(h) monitoring programs and the long-term effects of sewage discharge on the marine environment. This information will be used as a basis for decisions regarding permit reissuance.

The purpose of this paper is to provide an overview of ODES. The structure of the data base and data entry procedures are described briefly. Next, the statistical and analytical tools available on ODES are described, and example applications are given. Finally, the potential uses of ODES in other near coastal environmental programs are discussed.

INTRODUCTION

Section 301(h) of the Clean Water Act of 1977 and its subsequent amendments provide for case-by-case modification of secondary treatment requirements for discharges to marine waters by publicly owned treatment works. The 301(h) program is administrated by the United States Environmental Protection Agency (U.S. EPA) through the issuance of modified National Pollutant Elimination System (NPDES) permits. These variances are granted only when the discharger can demonstrate that the discharged wastewater of less than secondary quality will not adversely affect the marine environment. For example, statutory criteria include the attainment of water quality which protects public water supplies, a balanced indigenous population of shellfish, fish and wildlife, and recreational activities. As of September 28, 1987, a total of 208 applications for modified discharge permits have been received. Final decisions have been made by the U.S. EPA on 182 applications including 48 variances from secondary treatment.

A 5-year 301(h) modified NPDES permit was issued to each of the variance recipients. During this permit period, the discharger must conduct a comprehensive monitoring program to demonstrate continued compliance with permit conditions. A wide variety of data may be collected as part of these monitoring programs including influent/effluent quality, receiving water quality, benthic infaunal and fish abundance, fish histopathology, bioaccumulation, bioassay, and sediment physical/chemical characteristics.

The data collected in individual monitoring programs will be used to determine continued permit compliance. The data base that is developed from all permitted discharges will also be used to evaluate the effectiveness of 301(h) monitoring programs as well as long-term effects of sewage discharges on the marine environment. This information will be used as a basis for permit reissuance decisions. The data base is therefore a cornerstone of the 301(h) program, and the analysis of these data represents a key program activity.

SYSTEM DESCRIPTION

The Ocean Data Evaluation System (ODES) is a user-friendly computer system that was developed for the U.S. EPA 301(h) discharge program for storing and analyzing the monitoring data. The primary goal of developing this system was to provide a decision support tool that would accommodate a wide range of users and provide powerful analytical tools for the evaluation of the types of data collected as part of marine environmental management programs.

An overview of the structure of ODES is shown in Figure 1. The system consists of three basic components: user interface, integrated data base, and analytical tools. ODES resides on an IBM 3090 computer at the U.S. EPA National Computer Center at Research Triangle Park, North Carolina. The user interface which has been developed makes the system easy to access and operate so that no previous knowledge of computer programming is required to operate the menu-driven system. Additionally, users can access ODES via a local telephone call from virtually anywhere in the U.S. using almost any type of microcomputer

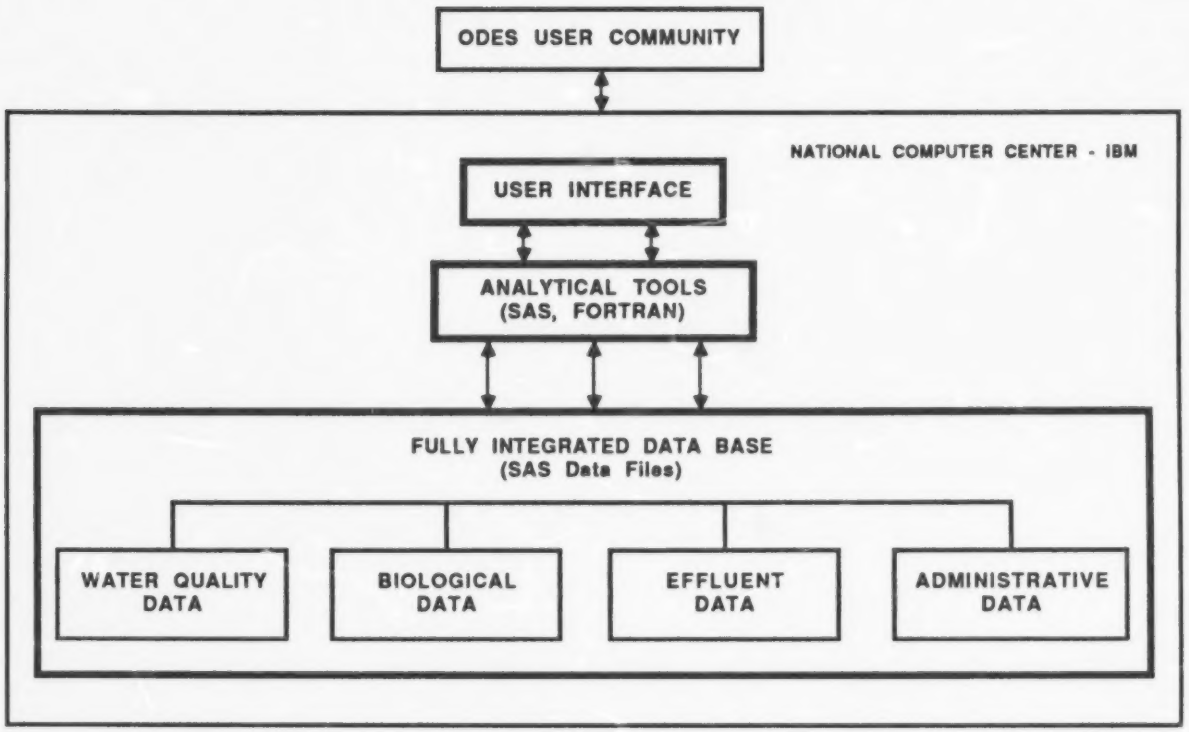


Figure 1. Structure of Ocean Data Evaluation System (ODES).

or mainframe computer terminal. At the same time, the system has been designed with built-in flexibility to allow advanced users direct access to the data base and the analytical tools. The ability has also been provided to directly access the data base and download information from the system for the production of reports or ad hoc analyses using analytical tools not available on ODES.

All data in ODES are stored as SAS (Statistical Analysis Software) data sets (SAS Institute Inc., 1982). This file structure allows simultaneous access to all data within the computerized system. Since the data are compiled from a large number of different monitoring programs and users, standardized data submission procedures have been developed to ensure accurate data entry. Key features of the data submission process include the use of standard formats for marine data developed by the National Oceanic and Atmospheric Administration's National Oceanographic Data Center (NODC) and an extensive quality assurance program. The quality assurance program includes the use of system software to produce data integrity reports and descriptive statistics for each file type. This information is then used by the technical staff in a detailed review of each data set submitted. Based on the results of this review, the ODES staff works closely with the data submitter to resolve errors, discrepancies, or omissions. Finally, a quality assurance report is produced which goes on-line with each data set.

The third component of the system is a set of statistical, graphical and modeling tools that enable the user to perform a wide variety of

statistical tests, conduct model simulations, summarize results, and produce reports and graphics. These tools are shown in Table 1 and include SAS statistical procedures (e.g., Analysis of Variance, Regression, and Correlation) as well as programs developed specifically for ODES. The latter group includes the Statistical Power Analysis and the Numerical Classification Tools that are described below. ODES also provides models for predicting particle deposition and organic sediment accumulation in the vicinity of sewage outfalls and estimating initial dilution characteristics of the effluent plume.

USE OF ODES

The major 301(h) program activities and the role of ODES as a decision support tool are depicted in Figure 2. The modified NPDES permit granted to the discharger requires the implementation of an approved monitoring program, and ODES provides a number of unique analytical techniques for the development and evaluation of alternative monitoring program designs. All data collected in the approved monitoring programs are then entered into the ODES data base. Subsequently, these data are analyzed using ODES statistical and graphical procedures to assist regulators in making program decisions.

As indicated in Figure 2, ODES is one of two major decision support tools developed for the 301(h) program. The other decision support tool is a set of guidance documents. In addition to three ODES User Guidance Documents, 16 monitoring guidance documents and four ODES Data Briefs have been developed as part of the 301(h) program. As shown in

TABLE 1. ODES ANALYTICAL AND DECISION-SUPPORT TOOLS

Tool Number	Tool Description
1	Plot of one variable over time
2	Plot of several variables over time (overlay)
3	Plot of means over time (with standards error bars)
4	Univariate means
5	Bar charts
10	T-test for replicated data
11	Wilcoxon/Mann-Whitney U test for replicated data
12	One-way analysis of variance for replicated data
13	Kruskal-Wallis K-sample location test for replicated data
14	Statistical power analysis
20	X-Y scatterplot with correlation statistics
21	Regression of a variable over time
30	Cluster analysis
51	Fish pathology contingency test
52	Calculation of an LC_{50} value for bioassay data
60	Plume models
61	DECAL (particulate deposition model)
70	National maps
71	Sectional maps
72	Study area maps
90	Mass loading
91	Nearfield pollutant loading
110	Retrieval of benthic survey data
111	Retrieval of auxiliary variables for benthic data
140	Retrieval of effluent data
141	Retrieval of auxiliary variables for effluent data
150	Retrieval of receiving water quality data
151	Retrieval of auxiliary variables for water quality data
160	Retrieval of grain size data
170	Retrieval of sediment pollutant data
180	Retrieval of trawl/seine data
181	Retrieval of auxiliary variables for trawl/seine data
190	Retrieval of bioassay data
275	Benthic indices
300	Summary of permits and monitoring plans
301	301(h) facilities summary
302	301(h) benthic survey data summary

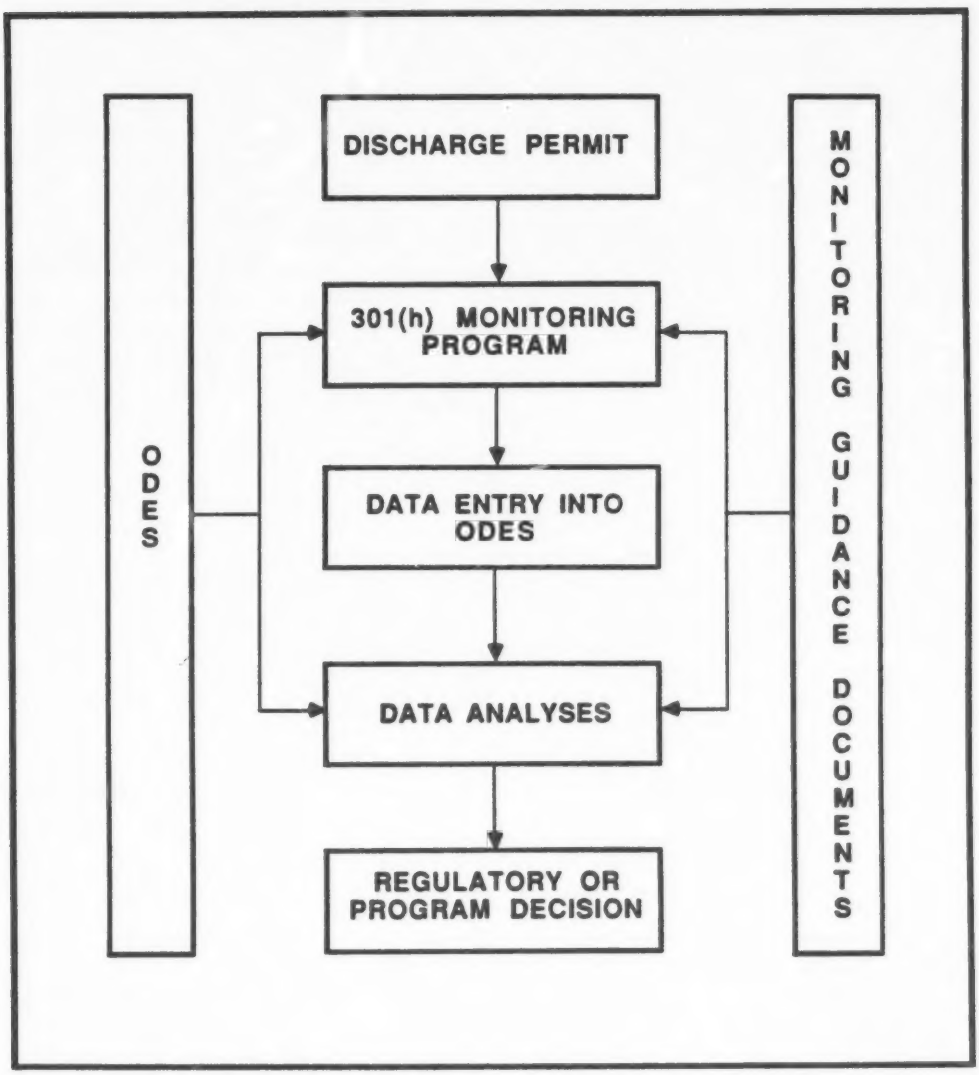


Figure 2. ODES as a Decision Support Tool.

the example applications of ODES described below, these documents complement ODES and address such issues as the selection of monitoring variables, analytical methods, and data analysis techniques.

EXAMPLE APPLICATIONS

Design of Environmental Monitoring Programs

The design of environmental monitoring programs involves five basic steps: development of program objectives, establishment of testable hypotheses, selection of analytical methods, evaluation of the minimum difference that can be detected between sampling locations or sampling intervals and the probability of detecting these differences, and finally the selection of the level of sampling effort. Several of the ODES tools (Table 1) can be used to assist in the design process, but the ODES Statistical Power Analysis Tool (Tool #14) was developed specifically to provide quantitative evaluations of the adequacy and practicability of monitoring programs. Without such an evaluation of alternative sampling strategies during the design phase, there is a risk of analyzing an excessive number of samples (with associated high costs) or of analyzing too few samples to detect real differences in measured values of the monitoring variable.

Previously, the use of power analyses to evaluate statistical procedures has been described in statistical texts (e.g., Scheffe, 1959; Winer, 1971), but the tools necessary to easily perform these analyses were not available. In addition to the statistical power analysis tool that has been developed for ODES, a complementary guidance document (Tetra Tech, 1987) has also been prepared that provides a review of the

basic concepts of hypothesis testing and statistical power analysis. This document also includes examples to demonstrate the use of the power analysis tool in making quantitative evaluations of expected monitoring program performance. These examples include the use of the power analysis tool to determine the power of the test (i.e., the probability of correctly rejecting a false null hypothesis) as a function of the minimum detectable difference between stations (Figure 3). Examples are also provided on the use of power analysis to determine the minimum detectable difference as a function of the number of replicate samples at each station.

Graphical Analyses

The data developed in individual monitoring programs will be subjected to various levels and types of analyses. The required analyses will take the form of developing and testing specific hypotheses. Typically, the sequence of activities associated with these analyses will begin with the use of ODES tools to qualitatively examine the relationship between the effluent discharge and biological and chemical variables. ODES provides a series of plotting and mapping tools that have been developed to facilitate these planned analyses. Additionally, an ODES Data Brief has been prepared to demonstrate the use of graphic and mapping tools in the analysis of the monitoring data.

Example applications of ODES tools to graphically display monitoring program results are presented in Figure 4. These tools provide the ability to easily access and plot information from single (Figure 4a)

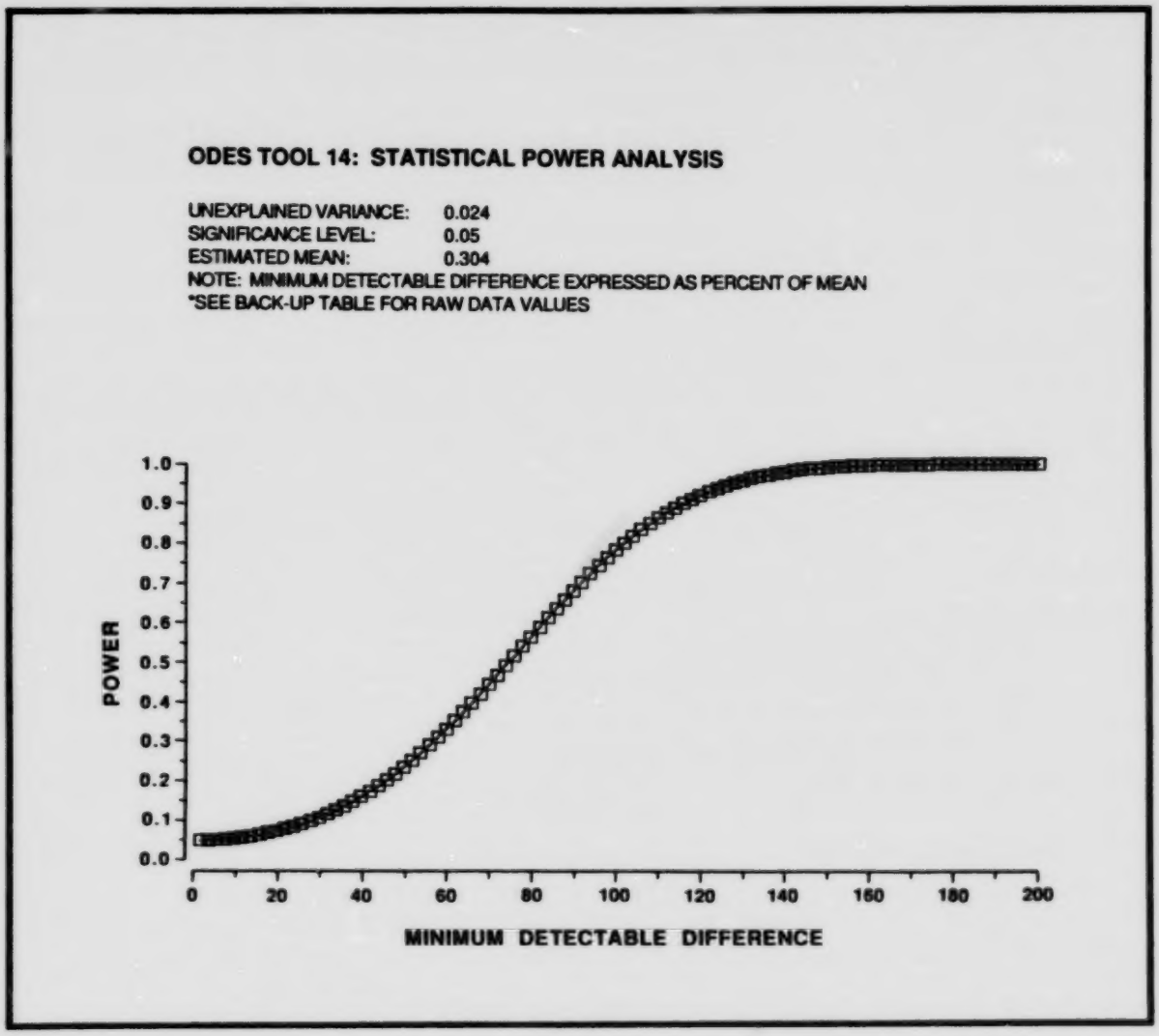


Figure 3. Example Output from ODES Statistical Power Analysis Tool.

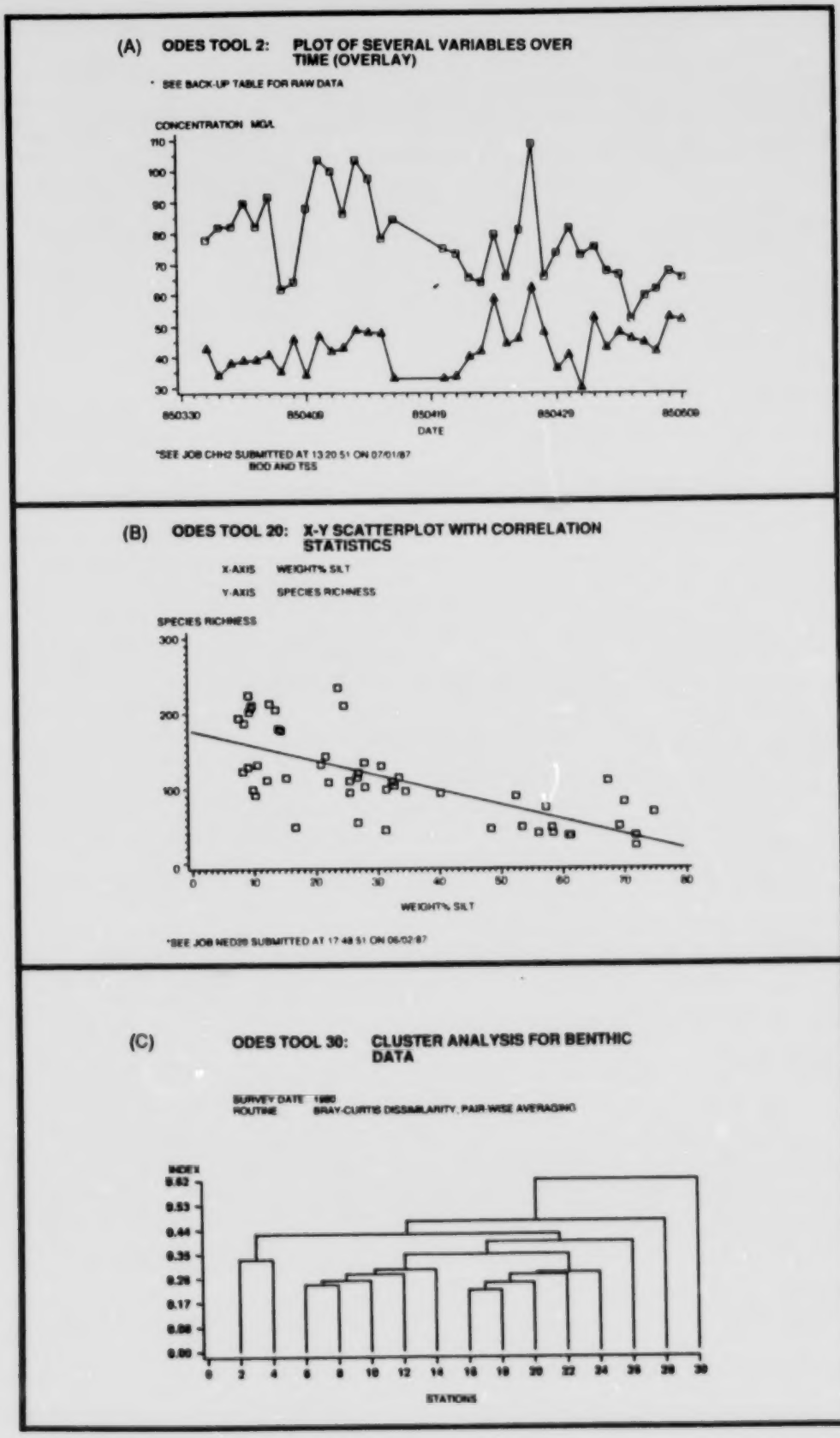


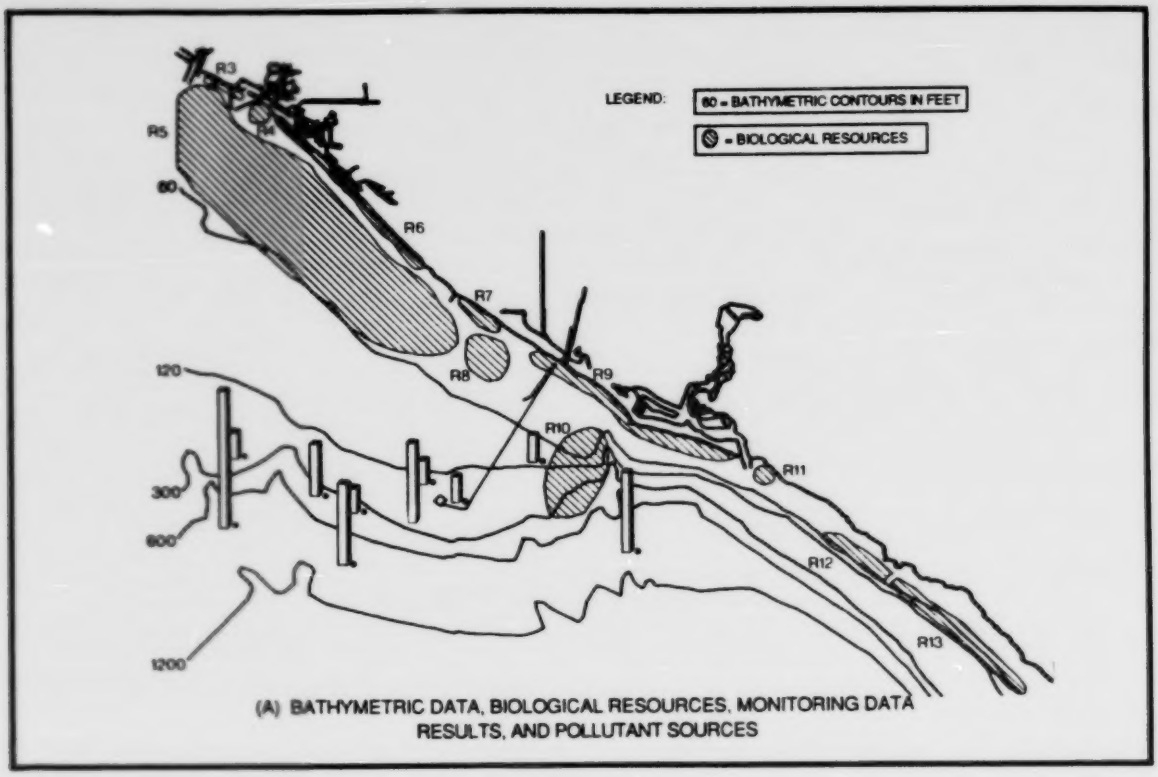
Figure 4. Example Output from ODES Graphic Tools.

or multiple database files (Figure 4b). Powerful heuristic tools such as cluster analysis (numerical classification) that can be used to distinguish groups of monitoring stations based on the similarity in biological community structure have also been developed and implemented on ODES (Figure 4c).

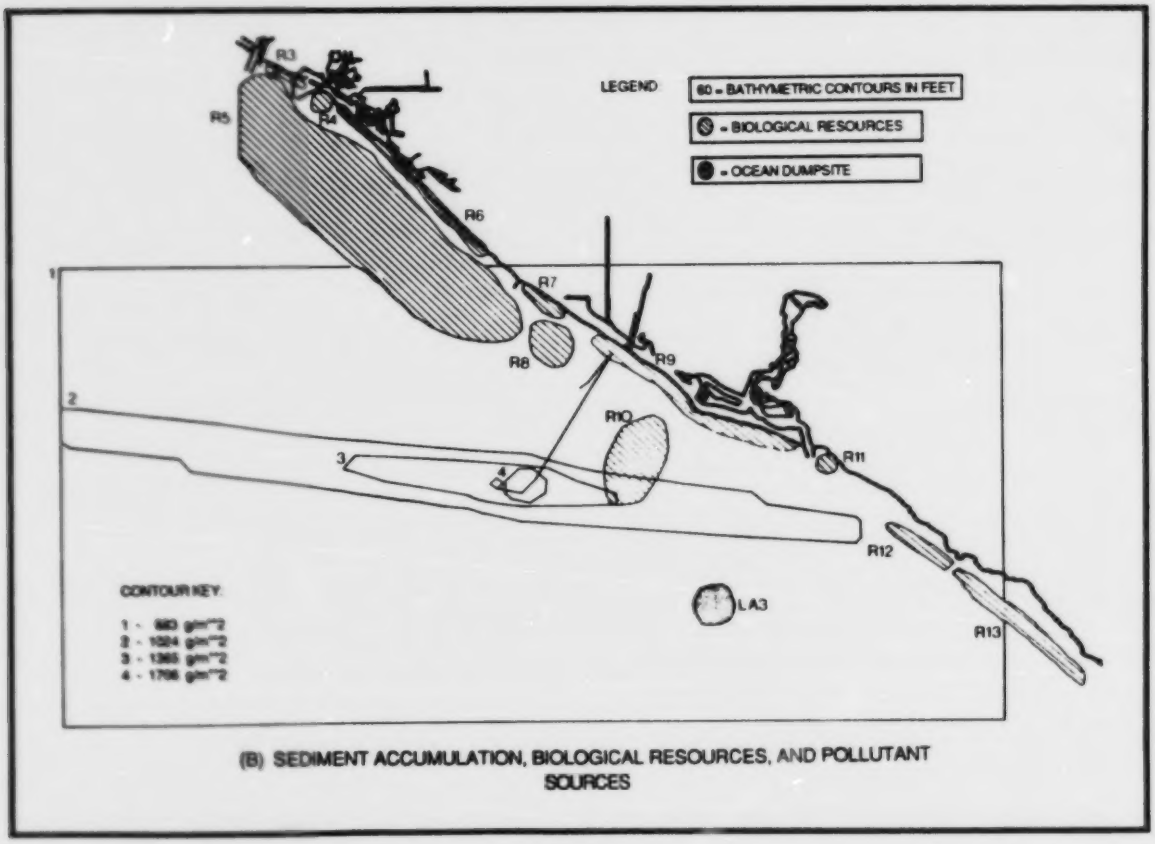
The mapping tools on ODES contain a series of national, regional and study-area maps and provide the user with the ability to overlay monitoring data as well as other data stored in the system onto these maps. Figure 5a shows an overlay of site bathymetry and biological resources as well as three-dimensional bar graphs of measured values of a monitoring variable on an example study-area map. Figure 5b shows how the mapping tool can also be used to display the results of the sediment accumulation model available on ODES (DECAL). These mapping capabilities coupled with the predictive model provide the user with the ability to evaluate how sediment accumulation might affect biological resources or what effect changes in mass loading might have on the receiving water. The results of such evaluations can then be used to guide regulatory decisions.

Statistical Analyses

ODES provides the user with easy access to the stored data as well as a set of statistical tools specifically selected for the types of data and analyses required in the 301(h) program (Table 1). Examples of these tools include: contingency table analyses for the evaluation of fish histopathology monitoring data; the LC_{xx} tool for the interpretation of quantal response (bioassay) data; and analysis of variance



(A) BATHYMETRIC DATA, BIOLOGICAL RESOURCES, MONITORING DATA RESULTS, AND POLLUTANT SOURCES



(B) SEDIMENT ACCUMULATION, BIOLOGICAL RESOURCES, AND POLLUTANT SOURCES

Figure 5. Example Study Area Map Created on ODES Showing Overlay Capabilities.

(ANOVA) and nonparametric tools for the evaluation of the statistical significance of observed differences in the values of monitoring variables.

During the initial phase of the 301(h) program, emphasis has been placed on the development of both statistical tools and monitoring guidance documents to support the design of monitoring programs. With the issuance of discharge permits and the implementation of 301(h) monitoring programs, current efforts are focused on the development of technical support documents and additional statistical tools to support data analysis activities.

OTHER APPLICATIONS OF ODES

While ODES was developed to support 301(h) program activities, the system has the capability of being used in other marine environmental management programs. For example, the ability of the system to accommodate data from other programs has been demonstrated by adding water quality, bioassay, and benthic infaunal sample data from ocean dumping monitoring programs to ODES. Additionally, several benefits could result from consolidating the monitoring data from different management programs. These benefits include the ability to quickly access a wider range of data as well as the ability to compare the results from different geographic areas or the influences of different pollutant sources. Perhaps the primary benefit from the existence of a central source of marine monitoring data, however, would be the ability to perform integrated analyses of the effects of several pollutant sources in order to make more informed environmental decisions. For example,

the computer-generated study area map presented in Figure 5b shows the location of both a permitted sewage discharge and a permitted ocean dump site. Consolidating the monitoring data from both disposal sites would allow a more comprehensive evaluation of potential environmental impacts.

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MODELS AND OBSERVATIONS OF INITIAL MIXING OF OCEAN-DUMPED WASTES: A STATUS REPORT

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ABSTRACT

The current U.S. Regulations for ocean disposal allow four hours for initial dilution of wastes to occur and specify several methods for estimating initial dilution with numerical models. Eleven numerical models and 33 field investigations were reviewed in a study sponsored by the U.S. Environmental Protection Agency to determine if there are data or other justification for the four-hour time period or the specified methods for estimating initial dilution. The four categories of wastes considered in the review include: 1) dredged material; 2) sewage sludge; 3) industrial wastes; and 4) drilling fluids.

Only 12 of the 33 field investigations reported estimates of initial dilution. Seven of these studies involved industrial waste disposal; the other five were of drilling mud disposal. Although more dredged material is dumped in the ocean than any other waste, none of the data sets that we examined are adequate for the calibration of the dredged material disposal models that are available. Time-histories show that the time rate of change of waste dilution in the ocean decreases abruptly within 10 to 30 minutes after discharge in most disposal operations. Dilutions ranged from 1,000:1 to 100,000:1 after 10 minutes in all cases. For industrial wastes dumped from barges and drilling fluids discharged from offshore platforms, initial dilution appears to be complete within 10 to 30 minutes after discharge.

Several numerical models are available that estimate initial dilutions of dense wastes such as dredged material and wastes discharged into the wakes of barges and drilling platforms. The coefficients required to simplify the mathematics in the models, however, are virtually unverified at prototype scales. Without validation of the model coefficients with field data or by comparisons with coefficients determined with large-scale physical models, the application of these models is limited to qualitative comparisons of initial dilutions of a few waste types in a limited range of oceanographic environments.

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INTRODUCTION

The current US regulations governing waste disposal in the ocean (40 CFR 227.29) allow four hours for ..."dispersion or diffusion of liquid, suspended particulate, and solid phases of a waste"... to occur. The regulations specify four methods for estimating initial mixing as defined therein. Numerical modeling is included in three of the methods. This study, sponsored by the U.S. Environmental Protection Agency, reviewed eleven numerical models and 33 field investigations to determine if there are observations and other sound justification for the four-hour time period or the methods for estimating initial dilutions with numerical models. Four categories of wastes were considered in the review; these include: 1) dredged material; 2) sewage sludge; 3) industrial wastes (acid and volatile); and 4) drilling fluids.

A detailed treatment of initial mixing concepts, field studies, waste dilution data, and numerical models is given by Ecker and Downing (1986). In this paper we provide an overview and summary of our findings. We compare, in tables with brief explanations, the characteristics and capabilities of numerical models that were available in 1986 and provide an evaluation of the data base for model validation and its bearing on the regulatory 4-hour time limit. In conclusion we comment on the limitations of the models and make some recommendations about the design of field studies to test them.

WASTE CHARACTERISTICS

The major categories of ocean-dumped wastes and some of their physical properties are summarized in Table 1. Dredged materials, industrial wastes, and sewage sludge comprise most of the wastes disposed in the ocean since the Second World War. The behavior of wastes during initial mixing depends largely on bulk density, total solids content, and solids density. In this paper, we divide the wastes into two general categories on the basis of solids content. Wastes with less than 10% solids are referred to as liquid wastes. They are about the

Table 1. Waste Characteristics

	Bulk Density (g/cm^3)	Total Solids Content (%)	Solids Density (g/cm^3)
Dredged Material	1.23 - 1.51	37 - 67	2.53 - 2.65
Drilling Fluid	1.15 - 2.30	10 - 70	1.3 - 4.5
Sewage Sludge	1.001 - 1.018	2.0 - 8.4	1.81
Acid-Iron Industrial Waste	1.025 - 1.2	1.7 - 12.2	Not Reported
Volatile Organic Industrial Waste	1.018 - 1.2	5.3 - 9.7	Not Reported

same density as sea water, and respond similarly to mixing processes in the ocean. When dumped at sea, liquid wastes are dispersed mainly in the surface layer by turbulence in the wake of the discharging vessel. Examples of liquid wastes are acid-iron and volatile-organic industrial wastes, and most sewage sludge. Dense wastes, those with more than 10% solids, set up strong convective disturbances when discharged in the ocean. They generally mix to greater depths than liquid wastes. Dense wastes include dredged materials, drilling fluids, and industrial wastes containing high concentrations of solids.

Acid-iron wastes contain as much as 8.5% acid and 10% dissolved iron by weight. Other constituents, suspended solids, heavy metals, and oil and grease, are present in quantities less than 2%. Reactions of acid-iron wastes with seawater produce settlable precipitates of iron. Volatile organic wastes contain as much as 6.3% solvents and degreasing solutions and as much as 5.5% oil, grease and suspended solids. The low-density organic materials in some industrial wastes spread on the sea surface where they are transported by the wind. Separation of precipitates and floatable compounds from waste plumes is of little consequence to initial mixing because these processes occur over a period of hours to days. Nonaqueous constituents of sewage

sludge include inorganic particles and organic materials such as proteins, amino acids, sugars, fats, and hydrocarbons.

Dredged materials consist of water, sediment (sand, silt, and clay), and fine-grained organic material. The liquid phase of dredged materials is comprised of interstitial water and seawater added during dredging and transport to the disposal area. The behavior of dredged material in the ocean depends primarily on the particle size distribution and water content of the material. These parameters are essential inputs to disposal models. Fine-grained sediments are generally cohesive and tend to remain clumped together during dredging and disposal. The density of clumps is similar to the in-situ density of the dredged material.

Drilling fluids are mixtures of materials for controlling wellhead pressures. Solids in drilling fluids include barite, bentonite, lignite, and a variety of other additives used to control pH, corrosion, foaming, friction, and other drilling parameters. Drilling fluids discharged into the ocean are water-based, and contain a wide variety of dissolved chemical additives to protect the geologic formation and the drilling equipment. Rock cuttings, with diameters ranging from microns to centimeters, and densities of about 2.6 g/cm^3, are also discharged at sea with waste drilling fluids.

PHYSICAL PROCESSES-INITIAL MIXING

At the outset of this review, the period of initial mixing was conceptually defined as the time required for the turbulence associated with the disposal operation and waste material to decay to predisposal levels. The time scale of the initial mixing phase is thus related to the rate of decay of turbulent energy rather than to the regulatory 4-hour time limit. After initial mixing is complete, passive transport and diffusion of the waste by oceanic turbulence, shear dispersion and advection occurs (Okubo et al., 1983). The rate of waste dilution during passive transport is much slower than during the initial mixing phase, and has a time scale of hours to weeks. The theory of initial

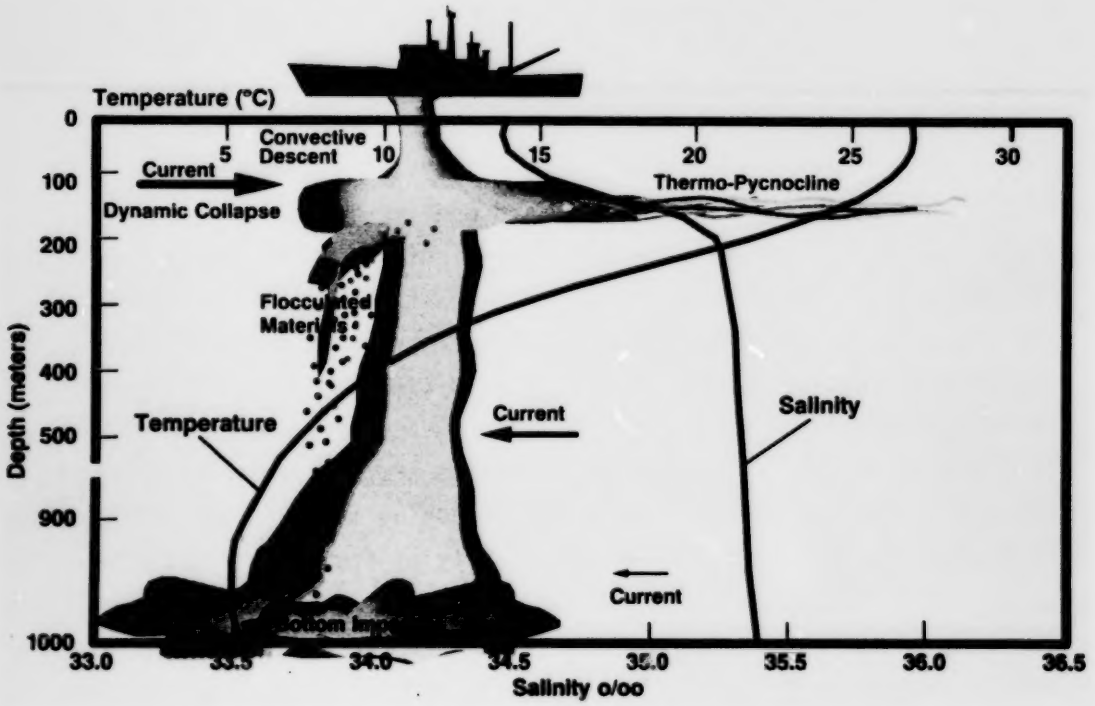

Figure 1. Disposal and Initial Mixing Processe for Dense Wastes

mixing is discussed by Koh and Chang (1973), Csanady (1973 and 1981), Fischer et al. (1979), Muellenhoff et al. (1985), and Roberts (1985).

Disposal methods and bulk density of wastes determine how they will behave during the initial mixing period. Figure 1 illustrates a disposal scenario in which a vessel, drifting with the surface current, is discharging a dense waste. In this situation, initial mixing results from the momentum and negative buoyancy of the dense material producing a rapid downward motion and turbulent disturbance in the water column (Pequegnat et al., 1978; Bokuniewicz, 1983). This process is called convective descent, and the descent velocity depends primarily on the bulk density of the waste. The velocity shear associated with the downward motion of the wastes generates turbulent vortices around the edge of the waste cloud causing entrainment of seawater into the waste field and dispersion of the waste material in the receiving water. The waste cloud accelerates to a maximum speed within a few meters of the surface and passes rapidly through the zone of wave and vessel-generated turbulence. Dredged materials dumped from barges settle at rates of 50 to 500 cm/s (USACE, 1978a and b; Gordon 1974). As water is entrained

into the waste cloud, it becomes less dense and slows down. Convective descent continues until the cloud impacts the bottom or the cloud density is reduced to that of the surrounding water. In either case, the cloud collapses and spreads horizontally in a phase called dynamic collapse. In shallow water, dredged materials can retain sufficient momentum to travel hundreds of meters as density flows after impact with the bottom. Collapse of the waste field can occur in the water column where there is a sharp gradient in the water density profile (Figure 1). Laboratory studies by Ozturgut (1982) suggest that retarded settling of particles in the Stoke's range at such a gradient is caused by the increased water viscosity rather than reduced negative buoyancy of the particles.

When dense wastes are discharged continuously from a vessel underway or a platform moored in a strong current, the turbulence of the wake contributes a very strong source of energy for initial mixing (Dames and Moore, 1978 and Ecomar, 1983). Since wake theory treats only basic shapes such as spheres, cylinders and discs, fully submerged and moving at constant speed in nonturbulent flow (Menkes et al., 1974), one must rely on very simple formulas or rules of thumb when dealing with mixing processes in wakes. Csanady (1981) suggests that the well-mixed part of the wake extends to three times the vessel draft and is 2.5 times the vessel beam in width. Clearly more research is needed in this very important area.

A third disposal scenario is the discharge of a liquid waste from a vessel underway. Csanady (1981) has shown by dimensional analysis that the inertia forces in the wake dominate the buoyancy forces associated with the wastes. Initial mixing is driven solely by the wake turbulence in this situation, and dilutions ranging from 1,000:1 to 20,000:1 have been observed (Delvigne, 1987; Frye and Williams, 1977). The effect of vessel speed was demonstrated by Ball and Reynolds (1976); their data showed that initial dilution increased exponentially with the speed of the discharging vessel. The International Maritime Organization formula, used in Europe for calculating initial dilutions,

includes the vessel velocity raised to the power, 1.4.

FIELD STUDIES

The locations of field studies are shown on Figure 2. Waste types are coded with symbols and the numbers in parenthesis reference the documentation for the studies cited at the end of the paper. Dredged materials (10 studies) and industrial wastes (10 studies) are the most extensively studied waste types. Drilling fluid ranks second (7 studies) and sewage sludge is third (4 studies). The studies are distributed widely in several oceanographic regimes. Only 12 of the 33 studies that we reviewed obtained data that are useful for making quantitative estimates of initial mixing.

Dredged Material Disposal

The studies of dredged material disposal involved monitoring instantaneous dumps of material from hopper dredges and bottom-dump

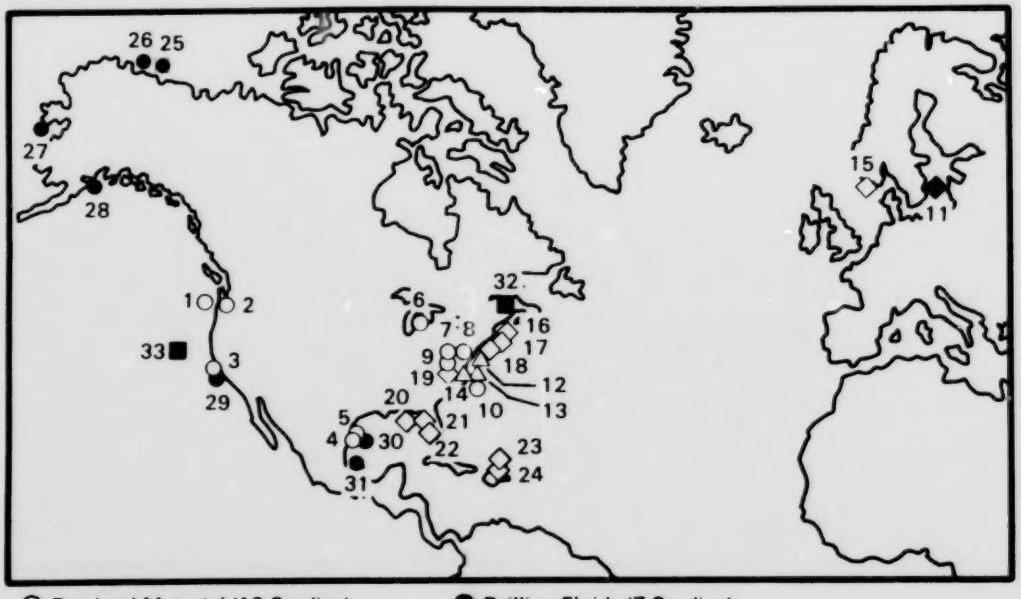

Figure 2. Map Showing the Locations, Waste Types, and key to References for Field Studies

barges. Post-disposal surveys indicate that 95% to 98% of the material settles to the bottom in the disposal site (Tavolaro, 1984). Collectively, the field studies conducted to date have documented: 1) the velocities and approximate dimensions of waste clouds during descent; 2) passive transport; and 3) the impact and distribution of the material on the bottom. Descent velocities of material discharged into Puget Sound, Washington reached 180 cm/s, and after impact with the bottom, material spread laterally at speeds of 20 to 36 cm/s (USACE, 1978b). Turbidity clouds near the bottom from dumps in Long Island Sound were initially 2 to 5 m thick in the impact area (Gordon, 1974). Within 10 to 30 min of the discharge, turbidity at the disposal site returned to near background levels.

Turbidity clouds of dredged material have been tracked with acoustic profilers (Data Sonics, 1983; Tsai and Proni, 1985) and side scan sonar (USACE, 1986). These methods provide an acoustic picture of the waste cloud in the mixing zone. At deep water sites, acoustic surveys can guide water sampling efforts and reveal secondary waste clouds that develop after initial mixing is complete. Water samples for calibration and computer deconvolution of the acoustic data are required to obtain quantitative information on waste concentrations. In 1987, the equipment to acquire the data and the technology to process it were not generally accessible to enforcement agencies and waste management groups.

Other important aspects of initial mixing that are simulated in numerical models of dredged material disposal, such as water entrainment, differential settling of coarse and fine particles, and flow dynamics in the waste cloud, have not been observed in the field. The lack of quantitative field data on instantaneous dumps of dredged material results mainly from the limitations of conventional sampling method and equipment. Also, since wastes in the water column represent only a small fraction of the total amount of wastes dumped, the mass balance must be confirmed by precision bathymetric surveys and bottom sampling; these are very difficult to do in deep water. For these reasons, none of the studies that we reviewed produced the data

necessary to verify or calibrate a comprehensive numerical model of initial mixing of dredged material.

Industrial Wastes and Sewage Sludge

Two approaches have been used to monitor initial dilution of industrial waste plumes. In the North Sea, a measurement system, moored to the bottom, was used to monitor pH, current velocity, temperature and salinity (Delvigne, 1987). The vessel discharged wastes along lines across the current upstream of the monitoring equipment which were profiled as they advected past the instrumentation. Plume tracking and precise maneuvering of a survey vessel was not required since the plume position was determined from current data. Another approach is to tow sampling equipment through a waste plume repeatedly to acquire tracer data and water samples as the plume ages. Frye and Williams (1977) characterized mixing of ocean-dumped wastes under best (rough sea states) and worst (calm) environmental conditions in this way. Their estimates of initial dilutions indicated that oceanographic conditions had no significant influence on initial mixing of the waste.

Drilling Fluid Disposal

Drilling fluid dispersion studies have provided the most informative data on initial mixing. Success in these studies has resulted mainly from innovative sampling equipment, helicopter operations, sediment traps and the great advantage of a waste source that remains in a fixed position. The studies covered a wide range of oceanic regimes including: the Beaufort Sea, summer and winter conditions; Norton Sound, Alaska, a high energy coastal regime; Cook Inlet, Alaska, a well-mixed estuary with very high current velocities; the Gulf of Mexico, a low energy environment; and two sites off California that are exposed to open-ocean swell (Figure 2). Dilutions of wastes measured within 10 minutes of discharge in these studies range from 1,000:1 to 1,000,000:1. Although the variance of current speed, an indication of the relative energy available for mixing in these areas, varied by a factor of 50, the initial dilution of wastes did not correlate with

oceanographic conditions in the expected way. Waste concentrations decreased faster in the low-energy environment. Sediment trap data from the Tanner Bank study suggest that 93 to 95% of the solids settles to the bottom inside the initial mixing zone.

In several studies, helicopters were used for the profiling of water quality parameters and sample acquisition. This technique provided virtually synoptic coverage of the plume. Sediment traps were also used in several of the studies to estimate the portion of material that settled from the water column. With the sediment traps, it was possible to make a mass balance of the material and to account for the partitioning of the waste between the water column and bottom deposition.

The 4-hour Time Limit

Twelve time histories of dilution, representative of two waste types, were developed with the field data to identify trends relevant to the 4-hour time limit. Five of these are shown on Figure 3 which represent the range of observations. For time periods of 10 to 100 min after discharge, dilutions vary by about three orders of magnitude. Within 10 min of discharge a minimum dilution of 1,000:1 is obtained and dilutions exceeding 100,000:1 are frequently achieved. A common trend of the time-history curves is the change in slope that occurs within 10 min of discharge. The slope and inflection point on individual curves is to some degree an artifact of the sampling schedule, however, the curves indicate that for a variety of oceanographic conditions, waste types, and disposal methods the most rapid mixing occurs within 10 to 30 min after waste discharge. We suggest that this range of times exceeds the time required for the decay of turbulence associated with drilling fluid and barged waste disposal. Because there is no comparable data from dredged material or sewage sludge, we can not extend this interpretation to the disposal of these wastes. It seems unlikely from similarity arguments, however, that the initial mixing period for other wastes, under similar oceanographic conditions, would be much different from those reported here.

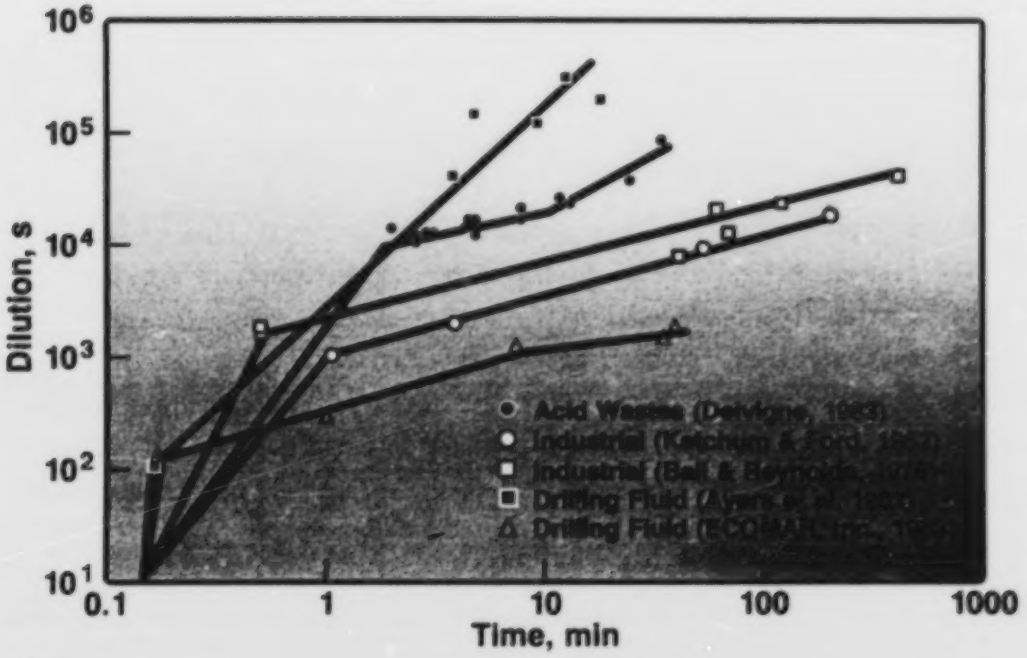

Figure 3. Time Histories of Initial Dilution

NUMERICAL MODELS

The features of the numerical models we reviewed are summarized in Table 2. The models are characterized on the basis of class (numerical or analytical); spatial domain; time dependency of input parameters (steady and transient); and method of validation. The division of the models into analytical and numerical classes is somewhat subjective because none of them use the state-of-the-art numerical methods that are routine in current hydrodynamic models. The intent of the classes is to indicate the degree to which physical processes are parameterized in the computer codes. The models classified as analytical rely on highly parameterized and empirical equations for calculations; models in the numerical class have formulas more closely linked to theory. Models which compute the horizontal variation of quantities over water depth are classified 3D. The remaining models have horizontal domains (2DH) or include water depth and one horizontal dimension (2DV).

Table 2. Features of the Models

Models	Class		Domain			Validation		
	Analytical	Numerical		Transient	Steady State	Field Data	Lab Data	Other Models
Barged Waste Disposal Model (Koh-Chang, 1973)		●	3D	●		●	●	
Modified Koh-Chang Model (Brandsma-Divoky, 1976)		●	3D	●		●	●	
Modified Koh-Chang Model (Bowers-Goldblatt, 1978)		●	3D	●			●	
Modified Koh-Chang Model (Wu-Leung, 1982)		●	3D	◐				●
EXXON Mud Discharge Model (Brandsma et al., 1983)		●	3D	●				
MIT Mass. Bay Model (Christodoulou et al., 1974)	●		3D		●			
CERCLA Type-A Assess. Model (Econ. Analysis-ASA, 1986)	●		3D	●				
Deep Sea Mining Model (Lavelle et al., 1981)	●		2DV		●	●		
106-Mile Dumpsite Model (Walker et al., 1984)		●	2DH	●				
Krishnappan Model (Krishnappan, 1975)		●	2DH	●			●	●

3D - Three dimensional
2DV - Two dimensional, one dimension is vertical
2DV - Two dimensional, both dimensions are horizontal

Validation refers to the method for checking coefficients in the model codes. The most robust methods include field observations; the least robust validations compare the results from one model to those from another, and use one set of results as a "standard" for reality. Validation with laboratory data implies that the selection of model coefficients is based on studies of specific processes with small-scale physical models.

The Koh-Chang Model and Its Descendants

Koh and Chang developed the first model of dense waste disposal that included convective decent, dynamic collapse and passive transport (Figure 1), for the U. S. Environmental Protection Agency in 1973. The model is capable of handling three discharge options and considers both the near- and far-field transport of wastes. The Koh-Chang Model

(hereafter, K-CM) treats the descent of waste material as a dense liquid cloud, using the equations for conservation of mass, momentum, buoyancy, and vorticity. The environmental inputs to the model include water depth and density structure and currents all of which are fixed over the model domain during a simulation. Waste characteristics included in the formulation are: 1) bulk density, 2) initial source radius and discharge velocity; and 3) the solids densities, fall velocities, and concentrations for up to eight solids components.

In the model, waste descends through the water column in a hemispherical cloud due to its momentum and negative buoyancy. A vortex ring forms around the cloud during descent causing mixing and dilution of the waste due to the entrainment of water into the cloud. Entrainment in the K-CM is assumed to be proportional to the velocity difference between the receiving water and the waste cloud and to its frontal area. Settling of solids during descent is a function of: 1) the ratio of the cloud velocity and the particle fall velocities; 2) the concentrations of solid components in the cloud; and (3) the total solids content. The variation of cloud mass with time is a function of water entrainment balanced by losses of particles from the cloud by settling. When the density of the cloud equals the ambient density, or when the cloud impacts the bottom, it collapses and spreads out laterally under the influence of gravity, bottom slope, fluid drag and entrainment.

Several coefficients and parameters are in the K-CM formulation for the descent and collapse of the waste cloud. They include coefficients in the equations for computing: fluid drag and entrainment, plume radius and density gradients, and particle settling. The coefficients for entrainment, drag, and particle settling are of great importance in the initial mixing computation. They determine the rate of water entrainment, growth and density of the cloud as well as the waste concentrations left behind in the water column. Laboratory and field data for estimating these coefficients for different environmental conditions, waste characteristics, and disposal methods are sparse at best. Johanson et al. (1976) showed by sensitivity analyses that the K-CM is insensitive to the solids content of the waste. Simulations with a

range of solids contents all resulted in about the same amount of waste deposition on the bottom. In laboratory studies, JBF Scientific Corporation (1975) found that dredged materials with greater than 50% solids, fall in cohesive clumps with very little dispersion of fine particles in the water column. Fine-grained materials with less than 25 to 30% solids settle as a dense fluid with considerable dispersion of fine particles into the water column. Use of a modified version of the **K-CM** for the Alcatraz Disposal Site in California resulted in gross underpredictions of bottom deposition and the eventual need to dredge the disposal site. The problem was traced to the treatment of sediment cohesion in the model. This sort of limitation bears on environmental problems as well. For example, if the model incorrectly partitions the fine sediment and adsorbed toxic substances of a contaminated dredged material between the suspended and bottom-deposited fractions, the fate of the toxic materials will be erroneously predicted.

Many modifications to the Koh-Chang model have been made since 1973. The characteristics of other models based on the K-CM and some additional ones that we reviewed are summarized in Table 3. Brandsma and Divoky (1976) added new force terms to the algorithm for dynamic collapse and a check for mass balance in the waste cloud. They modified the code for computing the distribution of material in the cloud, and added formulas for the fall velocity of cohesive material in the water column. Bowers and Goldenblatt (1978) further modified the model to include the effects of sediment cohesion and moisture content. They used laboratory experiments to develop better formulations for the entrainment and drag coefficients. The Exxon Mud Discharge Model (Brandsma et al., 1983 and O'Reilly et al., 1988) includes: 1) the effects of surface processes, such as wind waves; 2) enhanced settling, based on the concentration of solids in the water column; and 3) diffusion coefficients based on oceanographic conditions.

Other Models

The other models, such as the MIT Massachusetts Bay Model (Christodoulou et al., 1974), and the Krishnappan Model (Krishnappan, 1975) consider

some aspects of initial mixing of dense wastes. Although the MIT Massachusetts Bay model is primarily for studies of long-term dispersion, it does consider settling of waste material in general, and flocculation processes for fine cohesive particles in particular.

Table 3. Comparison of Model Characteristics

	Modeled Processes					Disposal Method			Waste Type		Environmental Conditions			
Models	Descent	Collapse	Bottom Impact	Wake Turb.	Passive	Slug	Continuous	Pipe Disch.	Liquid Waste	Dense Waste	Water Depth	Water Density	Current	Wave
Barged Waste Disposal Model	●	●	●	●	●	●	●	●	●	●	●	●	●	
Modified Koh-Chang Model	●	●	●	●	●	●	●	●	●	●	●	●	●	
Modified Koh-Chang Model	●	●	●	●	●	●	●	●	●	●	●	●	●	
Modified Koh-Chang Model	●	●	●	●	●	●	●	●	●	●	●	●	●	●
EXXON Mud Discharge Model	●	●	●	●	●			●	●	●	●	●	●	●
MIT Mass. Bay Model	●				●		●			●	●		●	
CERCLA Type-A Assess. Model	●				●	●			●			●	●	
Deep Sea Mining Model								●		●			●	
106-Mile Dumpsite Model				●	●	●			●		●		●	
Krishnappan Model	●					●				●	●		●	

* Refer to Table 2 for citations.

The 106-Mile Dumpsite model (Walker et al., 1984), CERCLA Type A Damage Assessment model (Economic Analysis and ASA, 1986) and Lavelle Deepsea Mining model (Lavelle et al., 1981), however, do not consider initial mixing of dense wastes. The 106-Mile Dumpsite model does include a treatment of wake turbulence based on vessel characteristics and the aforementioned rule of thumb. The EPA Model (EPA, 1984) was developed for site-specific investigations of dredged material disposal sites off the coast of Puerto Rico. The model consists of a variety of very simple and highly empirical formulas for calculating dilution during convective descent, passive transport and diffusion, and particle settling. The code has not been validated.

CONCLUSIONS AND RECOMMENDATIONS

Based on the field data reviewed for this study, initial mixing of liquid wastes discharged into the wake of a moving vessel and dense wastes discharged into the wake of a platform moored in a current is complete 10 to 30 minutes after discharge. This range of times probably exceeds the upper limit of time required for the decay of turbulence generated by the waste disposal operation. The regulatory 4-hour time limit therefore appears not to be linked to the observed physical mixing processes of these wastes.

The limited observations of key initial mixing processes from past field studies resulted from two main problems. The first one is a planning problem. Several studies that we reviewed attempted to achieve several major objectives with inadequate resources; the results reflect the misallocation of effort. The second problem relates to execution and sampling methodologies. With regards to the second problem, we recommend consideration of the following items when planning studies of waste disposal with emphasis on initial dilution:

1) Precise, real-time positioning of water-sampling equipment and sensors ($\pm$ 3 m, horizontal and $\pm$ 1 m, vertical);

2) Real-time determinations of the plume position;

3) A dense sampling grid with a minimum of ten levels in the depth of the plume and ten profiles across the plume;

4) A well-coordinated sampling team using equipment capable of acquiring several hundred measurements or samples in a twenty-minute period;

5) Incorporation of a method for determining a mass balance of the waste after initial mixing is complete.

Models of waste disposal rely heavily on coefficients that have not been validated at prototype scales. These coefficients need to be verified with additional field data or studies of physical mixing processes in large-scale physical models. Until this research is done, the application of numerical models of waste disposal will be limited to qualitative assessments of the behavior of a few waste types in a limited range of oceanographic environments.

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A Decision Framework for Site Management and Site Monitoring at Ocean Dumping Sites

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ABSTRACT

We propose a decision tree for site management and a conceptual framework for monitoring of ocean discharge permits. Site management and monitoring are organized according to the following principles: monitoring programs are driven by the Ocean Dumping Regulations; monitoring programs comprise assessment of permit compliance and possible impacts; site and waste characteristics form the technical foundation of the monitoring plan and determine the prediction of potential impacts; specific null hypotheses must be formulated based on predicted impacts; and these hypotheses should be organized into a tiered approach for efficient and cost-effective monitoring.

To demonstrate the effectiveness of this approach, we present sample hypotheses, methods to test the hypotheses, and management actions associated with acceptance or refutation of each hypothesis.

BACKGROUND

Recent discussions of monitoring strategies (Bernstein and Smith, 1986; Fredette et al., 1986; Wolfe and O'Connor, 1986) have emphasized that effective monitoring programs should focus on clear questions and testable null hypotheses; contain statistical rigor, including a strong sampling design and an ability to detect changes; and use baseline and available data

Clear questions and null hypotheses direct monitoring programs so that effort is not wasted by measuring variables that will not yield useful results. Clear questions can be used to focus monitoring on activities that will yield results that can then be used to manage the site. A monitoring program that does not focus on such questions may generate large quantities of data that can be

used to characterize the site, but can not be used to determine whether dumping has impacted the marine environment.

Formulating clear questions alone, however, is not sufficient to ensure an effective monitoring program. Statistical considerations must be part of the monitoring design to ensure that the program yields results that can be easily interpreted. For example, attention to statistical issues will make certain that enough samples are taken to be able to distinguish impacts from natural variability.

Baseline data are necessary for monitoring and evaluating potential impacts of ocean dumping because it is the change from baseline conditions that is being investigated. For example, chemical determination of known constituents of the materials being disposed of at the site would not be valuable without also knowing if those constituents were present before dumping began. Baseline data are necessary for all monitoring programs because these data provide information on natural variability of parameters that will be measured. An understanding of natural variability is necessary for designing a program that will generate data suitable for detecting changes from background conditions.

Currently available data can provide an easy focus for the monitoring program and prevent the collection of redundant or meaningless data. Many sites that may be used for ocean disposal, even offshore sites, have been the subject of environmental studies or monitoring programs. Use of appropriate data generated from those programs provides a realistic starting point for a monitoring program and may aid in interpreting monitoring results. In fact, if well-characterized materials are disposed at a well-studied site, an extensive monitoring effort may not be necessary.

In this document, we will address the characteristics of an effective monitoring program by presenting a stepwise approach to developing monitoring programs. The following steps are included:

1. Develop a conceptual framework for the program.
2. State the objectives of the program.
3. Develop null hypotheses.
4. Group the null hypotheses into "tiers."
5. Select parameters and the associated methods to collect data on those parameters.
6. Describe the variability of those parameters within the natural system.
7. Develop a monitoring program design by selecting the number of stations and replicates that will allow detection of changes in parameter values of significance to site managers.

If these steps are followed, data generated by the monitoring program will be complete, defensible, and useful for making management decisions.

CONCEPTUAL FRAMEWORK OF THE MONITORING PROGRAM

To develop a conceptual framework for monitoring, existing information about the characteristics of the dumpsite and the waste is used to approximate the potential for effects of waste disposal. The actual monitoring that may be necessary for a site is determined on a case-by-case basis. The nature and extent of monitoring depends on the the characteristics of both the dumpsite and the wastes to be disposed.

Waste characteristics guide the conditions of the permit and the design of the monitoring program. Whether constituents of a waste are soluble and, if not, whether the particles remain in suspension or settle will to some

extent determine what impacts will occur, and will therefore guide selection of monitoring methods and location of sampling stations. Whether a waste is inert or chemically or biologically active will also determine what sort of monitoring is necessary.

For example, inert wastes would primarily have physical impacts on the marine environment, and the monitoring program should be designed to focus on these impacts. In contrast, wastes that are known to be toxic or that may interact chemically or biologically with the environment require more extensive biological monitoring. The scope of monitoring programs will vary considerably: less intensive or frequent monitoring will be necessary for small amounts of inert material dumped at sites well removed from significant marine resources; and frequent or detailed monitoring will be necessary when chemically active materials are continually dumped near significant marine resources.

Two additional factors should be considered in designing monitoring programs. First, the amount of information already available may affect the monitoring requirements. For example, a material whose effects are well known would require less monitoring than one for which information is less extensive (because it has not been dumped before) or less definitive (because results have been ambiguous). Second, the stage of the monitoring program may be a factor. For example, the initial stage may require intensive monitoring to assess dumping effects, whereas later stages may permit a reduction in effort if the initial data verifies that dumping has no adverse effects.

OBJECTIVES OF MONITORING

All dumpsite monitoring programs are ultimately designed to ensure that dumping of wastes in the ocean does not adversely affect human health or the marine environment. This intent can be broken down into two categories corresponding to monitoring objectives defined in the Ocean Dumping Regulations; that is, to provide relevant information needed (1) to evaluate compliance with permit conditions and (2) to determine the impacts of dumping.

Permits issued for ocean dumping establish operational conditions that must be complied with by the permittee. These conditions may include specific disposal rates or restrictions on the time of dumping. In general, compliance with permit conditions is assumed to be sufficient to protect human health and the marine environment (i.e., to ensure no impact). Monitoring activities, however, are designed not only to verify that permit conditions are met during dumping activities, but also to verify the assumption that the conditions set by the permit were sufficient to ensure no impact. The mechanism for this verification is an iterative process of developing predictions of impact based largely on information gained through the site designation process, formulating null hypotheses based on these predictions, and verifying the predictions by testing the null hypotheses. In this way, the monitoring program is responsive to monitoring results, and collection of data is focused on acquiring information needed to make sound management decisions.

DEVELOPMENT OF NULL HYPOTHESES

Monitoring programs can be designed most effectively if they borrow the concept of null hypothesis testing from scientific experimentation. Implicit in the concept of an experiment is a question or null hypothesis that is being evaluated. A monitoring program focused on answering specific questions or testing null hypotheses concerning compliance with permit conditions and potential impacts of disposal of wastes at sea will be designed quite differently than a program that is viewed as simple data collection (Green, 1979).

Null hypotheses should be based on predictions of impact developed from information on site and waste characteristics. This type of monitoring design will ensure that field and laboratory studies are designed to verify null hypotheses. The experimental design approach requires continually asking the question, "Will measuring this variable allow me to make a management decision about the permitted disposal?". By answering this question the link between predicted impacts, null hypotheses, monitoring, and management becomes more explicit.

The kinds of null hypotheses to be developed can be grouped into six categories: waste characterization, disposal operations, nearfield and farfield fate, and short-term and long-term effects. Null hypotheses concerning waste characterization, disposal operations, and some nearfield fate issues (e.g., compliance with Limiting Permissible Concentrations) address permit compliance and are used in determining what null hypotheses should be tested to assess potential impacts. Null hypotheses concerning nearfield and farfield fate and transport, and short- and long-term effects address impact assessment.

"Nearfield," "farfield," "short-term," and "long-term" have been variously defined in ocean disposal programs in the past. The distinction between short- and long-term is

often difficult to define. The Ocean Dumping Regulations define initial mixing as being within four hours after dumping. In reality, this process can take as little as 10 minutes. For this document, we have used the following definitions:

- Nearfield--within the disposal site boundary.
- Farfield--outside the disposal site boundary.
- Short-term--four hours to one day after dumping.
- Long-term--greater than one day after dumping.

Ideally, the null hypotheses to be tested would perfectly correspond with the six impact criteria used to evaluate the status of ocean dumping at a site [40 CFR 228.11(b)]. In practice it may be very difficult to determine, for instance, if "there are statistically significant decreases in the populations of valuable commercial or recreational species" (40 CFR 228.11) as the result of the use of a particular ocean dumpsite.

TIERED APPROACH TO MONITORING

Once null hypotheses have been developed, the most effective strategy for assessing waste disposal impacts is to use a tiered monitoring approach (Zeller and Wastler, 1986). Such an approach will generate only information necessary for decision making. Explicit monitoring objectives, endpoints, and null hypotheses are organized within a hierarchy of tiers. The objectives in each tier focus on regulatory or environmental protection endpoints against which measured effects can be compared. Thus, each tier has a set of null hypotheses that can be tested in the field.

Activities involved in waste characterization and waste disposal are included in the first tier. Second-tier

monitoring activities test null hypotheses dealing with short-term transport, fate, and biological effects of ocean dumping. The third tier tests null hypotheses dealing with long-term fate and effects of dumped wastes. Additional tiers may be required if long-term effects are detected. These tiers would assess impacts on fisheries or other resources of commercial or intrinsic value. It is important to note that both the number of tiers and the relationships among tiers will be program specific, depending on site and waste characteristics.

The conceptual basis for a tiered approach to the monitoring program is that data collected in each tier are needed as the basis for the monitoring activities in the next tier. Thus, data collected in the early tiers on the chemical composition and properties of the waste and disposal operations are needed to choose the appropriate chemical parameters for measurement in the nearfield. Data from middle tiers on the composition, movement, and toxicity of waste in the nearfield are required to choose the best locations for measurements in the farfield and the most appropriate parameters to measure in later tier monitoring activities.

Another important attribute of the tiered approach to monitoring is that the outcome of the monitoring activities in each tier provides a technically sound basis for deciding whether specific types of monitoring activities are needed in the next tier. For example, if monitoring activities in an upper tier show that concentrations of metals in the disposal plume are diluted to background concentrations within the boundaries of the disposal site, then in lower monitoring tiers, those chemicals may not be looked for in the farfield outside the site boundaries.

The basic design of the tiered approach dictates that the environmental and program-related questions to be answered by the monitoring program become more complex as the program moves from early to later tiers (Zeller and

Wastler, 1986). The final monitoring tiers focus on long-term effects of ocean dumping on commercially or recreationally important marine fishery resources; on marine reptiles, birds, and mammals; on sensitive marine habitats and recreational beaches; or on other marine resources of social or economic value. Because complex, lower tier activities depend on upper tier results, upper tier studies may justify a decision not to conduct the lower tier studies. In other words, a tiered monitoring strategy assumes that acceptance of the null hypotheses in upper tiers will provide protection from the impacts cited in the regulations (40 CFR 228.11). In fact, for many sites, the absence of farfield/long-term impacts will be reliably predicted by nearfield/short-term activities, and thus no farfield/long-term activities will be required.

Once a null hypothesis has been verified within an acceptable range of uncertainty, related field studies may be reduced to a level that is sufficient to verify that the null hypothesis is not violated under varied waste characteristics and disposal conditions. Null hypotheses can also serve as decision points for moving from upper to lower tiers. Ideally, detailed studies in a lower tier will be conducted only when data show that an upper tiered null hypothesis is false, or if the results are uncertain.

Hypotheses must be framed by considering the statistical and logistical constraints on monitoring ocean dumping. Statistical considerations are necessary to ensure that the monitoring activities yield results that are meaningful and appropriate for hypothesis testing. Logistical considerations ensure that the monitoring design can actually be implemented.

Waste Characterization

Permits are issued based on specific physical, chemical, and biological characteristics of wastes. Waste characterization hypotheses (Figure 1) are designed to determine whether wastes comply with permit limits and regulations. Certain assumptions can be made about the initial dilution of the waste, and the maximum concentrations of waste constituents can be set to ensure that the waste will not exceed the LPCs after initial dilution, and hence result in toxicity. Waste characteristics, however, can vary dramatically over time. If the waste characteristics change significantly, the assumptions made in writing the permit may no longer be valid. Monitoring the waste composition will ensure that these permitted concentrations are not exceeded. These activities of the monitoring program are generally carried out as part of the permit compliance monitoring program performed by the permittee.

Disposal Operations

Permit conditions also determine the location and rates of discharge. Information about vessel characteristics and discharge rates is also essential to determine the initial dilution during dumping operations and whether the LPCs are met. Compliance with permit conditions specifying disposal operations can be tested through the use of automatic tamper-resistant or tamper-detectable devices to record the following navigational parameters: time, date, vessel position, and vessel speed. These activities are considered to be compliance activities and generally are the responsibility of the permittee.

Nearfield Fate

Some of the null hypotheses (Figure 2: NF-B, NF-D) concerning nearfield fate of the wastes disposed at sea are also related to monitoring permit compliance (i.e., they examine the assumption that LPCs are not exceeded in the disposal site four hours after dumping or at any time outside the site boundaries). Other null hypotheses are designed to determine whether the disposal operations impact the environment.

Formulation of these null hypotheses depends on the physical characteristics of the wastes. For liquid wastes, null hypotheses are directed towards determining the movement of the waste plume in the water column. For dense solid wastes, null hypotheses are directed towards determining whether the waste remains within the site boundaries.

Models exist, or can be formulated, to explain the behavior of some materials after dumping. When such models exist, nearfield null hypotheses can be stated in terms of testing or validating model predictions. Although model testing is not an objective of monitoring programs, verification of models could result in their increased use for site management.

Verification of initial dilution is difficult and expensive, requiring extensive arrays to collect water samples (Segar and Stamman, 1986). During the first year or two of the permit, it may be necessary to verify null hypotheses concerning rates of initial dilution under a wide range of oceanographic conditions. Unless it can be assured that waste characteristics do not change from one dump to another, these null hypotheses must be tested throughout the program.

The assessment of nearfield transport will be relevant in assessing farfield transport and fate. For instance, if a significant percentage of the particles sinks below the

seasonal or permanent pycnocline, long-term contamination of sediments and bioaccumulation by benthic animals may have to be assessed.

Short-Term Effects

Short-term effects in the nearfield are most likely to be detected while conducting short-term toxicity tests using endemic species (Figure 3: ST-C). The determination of acute toxicity of wastes proposed for ocean dumping, required as part of the permit application process, should demonstrate that no local species are more sensitive than the species used in the toxicity tests. Compliance with water quality criteria and LPC requirements will, therefore, usually be adequate for verifying predictions of short-term effects. Research at the physiological and subcellular levels is also under way on other indicators of short-term stress including adenylate energy charge, respiration rate, scope for growth, and genetic alterations. Null hypotheses based on these methods could be developed to determine sublethal effects of ocean-dumped wastes.

Farfield Fate

Null hypotheses (Figure 4) in this category are concerned with the long-term transport and degradation of the discharged material. For a soluble material, null hypotheses concerning farfield fate could include demonstration that waste constituents will not be transported to shores and beaches, critical areas, or marine sanctuaries [40 CFR 228.11(b)(1)]. Segar and Stamman (1986) argue that testing these null hypotheses would probably be linked to a nearfield measure of floatable materials. Besides transport of materials to these critical areas, farfield fate hypotheses are

concerned with determining whether waste constituents or by-products are found in toxic concentrations above normal, ambient values [40 CFR 228.11(b)(5)].

For materials that settle to the bottom, null hypotheses concerning farfield fate would be directed towards long-term deposition of waste solids outside the dumpsite boundaries to determine if major uses of the area are significantly impaired [40 CFR 228.11(b)(3)]. Detecting change in sediment concentrations of waste constituents is dependent on long time scales because of the potential for historic deposits of materials in the sediments to mask recent sedimentation. For this reason, many scientists have suggested the use of sediment traps to monitor sedimentation of waste solids over short time scales (Segar and Stamman, 1986; Rowe and Gardner, 1979).

Long-Term Effects

The Ocean Dumping Regulations indicate that ocean dumping should be reevaluated when dumping causes statistically significant decreases in the populations of valuable commercial and recreational species or of specific species of biota essential to the propagation of such species [40 CFR 228.11(b)(2)]. Some chronic population effects are possible at low exposure concentrations. However, because of the large variability in population abundances that occurs naturally over space and time, it may be difficult to detect effects related to disposal activities at the population level. For this reason, long-term effects should be monitored in later monitoring tiers only after it can be determined where effects are most likely to occur and which effects should be measured through short-term monitoring. As previously discussed, long-term effects will be determined only when justified by predictions of long-term effects based on short-term/nearfield results.

These null hypotheses (Figure 5) should also evaluate the concentrations of chemicals in tissues of indigenous organisms harvested by commercial and recreational fishermen [40 CFR 228.11(b)(4)]. Simulation models of some ocean dumping scenarios have indicated that the greatest risks are associated with bioaccumulation of lipophilic, organic compounds (e.g., Walker et al., 1987). The ongoing National Oceanic and Atmospheric Administration Status and Trends Program, including mussel watch and benthic surveillance programs, is addressing the question of long-term accumulation of contaminants in nearshore coastal waters; those programs may provide useful information about bioaccumulation of waste constituents. Long-term effects of waste disposal then would be monitored in indigenous populations demonstrating evidence of bioaccumulation of waste constituents.

Selection of Null Hypotheses

Selection of appropriate null hypotheses depends heavily on waste and site characteristics. Whether a waste is soluble, remains in suspension, or settles, along with information on physical processes and characteristics of the site, will influence the types of impacts that will be predicted. The selection of null hypotheses based on impact predictions is therefore dependent, to some extent, on these factors. Table 1 presents the relative importance of waste and site characteristics to the sample null hypotheses described in this section. For example, null hypotheses concerning water quality criteria in the nearfield (NF-B WQC) would be less important for dense, solid wastes such as dredged materials than for liquid, buoyant wastes such as sewage sludge because of the high probability that solid wastes will settle entirely with little effect in the water column. Similarly, farfield null hypotheses concerning the transport of liquid wastes

to shorelines (FF-C Shorelines) would be far more important at a coastal dumpsite than at an ocean dumpsite, given the relative distances of the two from the shoreline.

SELECTION OF PARAMETERS

Parameters chosen for monitoring should represent a direct linkage between the management concern and the hypothesis. Parameters are chosen with the following characteristics:

- Meaningful socially, economically, and environmentally.
- Sensitive to the impact.
- Relatively invariant in the unimpacted, control situation.
- Cost effective to monitor.

Parameters could be "meaningful" because of their inclusion in the impact criteria (40 CFR 228.11). For instance, it could be necessary to measure dissolved oxygen because there is a water quality standard for dissolved oxygen. Alternatively, parameters could be selected for their environmental importance and framed within a null hypothesis; for instance, a null hypothesis could state that a specific water quality criterion is not exceeded within the dumpsite after initial dilution.

Parameters may also be chosen for their description of waste fate and transport; for instance, they may address whether the waste settles after it is dumped. Some may be tracers rather than parameters that have demonstrable ecological significance. Dyes added to the waste material are an example of tracers. By tracing the movement of the waste material, sampling for biological impacts can be directed and performed more efficiently. Parameters measured to verify dispersion calculations should include

only those dissolved and particulate constituents whose concentration changes in the water column are highest due to dumping. Downcurrent dilution of one or two tracers should be measured. Possible changes in other parameters can then be calculated based on concentrations within the waste.

Quantification of the sources of variance, as identified earlier, will provide data to make choices based on statistical efficiency and cost effectiveness. Effective monitoring programs measure parameters that provide the most accurate and precise estimate of a mean value for the smallest sampling effort. This strategy will maximize information return per sampling effort. Selection criteria for parameters include the following:

- Parameters that are not currently present in the environment (presence/absence tests may be easier to perform statistically than quantitative evaluation).
- Parameters with inherently low variability. This will provide more stable (and tighter confidence limits on means) and more powerful and accurate tests of hypotheses.
- Parameters that are easily measured and expected to show maximum effects of dumping. This might be especially true for hypotheses about biological effects, for which the focus should be on organisms known or expected to bioaccumulate certain kinds of chemicals.

A sampling program must also distinguish between natural variability in selected parameters and actual effects of the dumping activities being monitored. Parameters may be chosen for measurement not to detect effects associated with waste disposal, but as indicators of the oceanographic conditions that might influence natural variability. These parameters may include temperature, salinity, and density.

DETERMINATION OF SOURCES OF VARIABILITY

Monitoring the impacts of ocean waste disposal on living marine resources of the ocean is made particularly difficult by the large spatial and temporal fluctuations in these parameters. Variability is defined here as any deviation among measurements that cannot be directly associated with the effects of waste disposal. As a result of this variability, monitoring programs must be designed to minimize this "signal-to-noise" problem. The variability must be described and quantified using baseline information to achieve cost-effective sampling and valid experimental designs.

Design of monitoring programs in oceanic waters must consider the natural spatial and temporal patterns of variability. These patterns are important in choosing the appropriate spatial and temporal scales for measurement. For instance, temporal variability can affect within-year and between-year variability in measurements. Sampling regimes can be controlled for time of year, or a stratified sampling design can be chosen. Use of yearly averages is rarely a good idea. In general, the most sensitive time for monitoring will be when mixing is minimized during summer because the shallow thermocline acts as a barrier to vertical mixing. If seasonal samples are taken, they should be part of a stratified sampling design.

Not all sources of variability are important on all spatial and temporal scales. Biological communities exist within implicit spatial and temporal scales. Samples collected on a specific temporal and spatial scale will reflect processes acting on that scale. The implications for monitoring plan design are that null hypotheses tested for nearfield and short-term effects must consider different kinds of variability than null hypotheses considering farfield and long-term effects. Variability must be determined for both background and dumping

conditions. For instance, a chemical that is below detection limits in background conditions will have an indeterminate mean and variance, but the variance could be quite high in the nearfield, short-term scale.

Besides spatial and temporal variability, monitoring programs must also evaluate the magnitude of sampling variability. Replicate variability can consist of field collection error and laboratory measurement error. Laboratory variability is error added by analytical techniques. Some measurement procedures for certain compounds may be quite variable, whereas others are precise and accurate. For approved EPA methods, the required level of precision associated with the technique is generally stated. Field error is variability added to measurement by field collection techniques, for instance, the types of containers and materials used to collect the samples.

DEVELOPMENT OF OPTIMAL SAMPLING AND ANALYSIS PLANS

The goal of plan optimization is to develop the most powerful hypotheses tests possible. The power of the hypotheses tests depends to a great extent on the number of times paired comparisons can be made between reference and surveillance locations. Therefore, it is important to plan monitoring activities to (1) maximize the number of paired comparisons that can be completed in a given time and (2) allocate samples to minimize the error and thereby increase the power of hypothesis tests with a given number of paired comparisons.

Optimization is achieved by balancing statistical, logistical, cost, and management considerations. Each consideration is discussed separately in this section.

In order to select the number and locations of monitoring stations, a statistical model for the monitoring program must be selected. This model could be either parametric or nonparametric. Parametric models range from

simple t-tests to highly stratified analyses of variance (ANOVA). In some instances, the assumptions necessary for ANOVAs (e.g., homogeneity of variances) will be violated, and it will be necessary to use nonparametric tests. For example, nonparametric tests would be needed if the background concentrations of a chemical to be dumped are below detection limits.

After choosing the statistical model to be used, it is possible to determine how many samples are necessary to detect a statistically significant change at a given detection level (e.g., Bernstein and Zalinski, 1986). To make such a determination, it is necessary to know the ambient mean and variance of the parameter of interest. The variances among stations, sampling replicates, and analytical replicates can be determined from a baseline sampling program, and the effect of varying the number of stations and replicates on total variance can be calculated. Determination of the number of samples is dependent on the desired "minimum detectable change," generally referred to in statistics texts as "minimum detection level" (a usage easily confused with chemical "detection limits"). The site manager will need to constantly evaluate the "managerial significance" of differences between reference stations and stations within the dumpsite and the necessary scale of the monitoring program.

In developing optimal sampling and analysis plans, field oceanographers also must be consulted for sampling designs that are not possible because of logistical constraints expected at sea. Practical constraints that must be taken into account includenavigation and station keeping, weather, time necessary to deploy sampling equipment, uncertainty associated with movement of water masses, and endurance of ship and crew.

All oceanographic monitoring surveys must include contingency plans to deal with each of the above

constraints, and the statistical design must be specified with knowledge of the changes that may be necessitated by the imposition of these constraints.

After considering the limitations that budget, time, and background variability place on the "minimum change" detected between reference areas and potentially impacted areas, site managers should reconsider whether the parameters they have chosen will yield useful information. Perhaps alternative parameters should be sampled. Alternatively, it might be necessary to reformulate the hypotheses into more easily tested questions. Most likely, developing a monitoring plan will require several iterations.

USE OF MONITORING RESULTS FOR SITE MANAGEMENT

Strategies for managing sites for disposal of ocean waste and for choosing management options should be determined on a case-by-case basis. Site and waste characteristics and permit conditions will affect the kinds of decisions a manager can make and must be considered for reasonable site management. Monitoring program results can be used by the site manager to decide on site designation, permitting, and future monitoring.

Figure 6 presents an idealized decision tree that uses information from the tiered monitoring strategy discussed previously. The essence of this decision process is a series of questions, the answers to which determine the options chosen by site managers. Rather than list a specific course of action for each scenario that may be encountered in managing waste disposal sites, this section discusses an example of a decision process that might be followed in evaluating management options. This process is considered in a generic sense and should be modified to suit the needs of specific sites.

A basic question to be asked in managing a

disposal site is "Do the wastes and disposal operations meet permit and site management requirements?" As has been discussed, specific disposal requirements are established in each permit, and compliance with these requirements is a condition for continued use of the site. Site management requirements are generally outlined in the Ocean Dumping Regulations and may be specifically stipulated in individual site designation regulations. If permit and site management requirements are not being met, then enforcement action may be required to bring disposal operations into compliance. Enforcement action can include levying fines, issuing administrative orders, or revoking permits after appropriate evaluation. Negotiations between the dumper(s) and the site manager may also be called for to revise existing permit conditions and site management requirements.

If monitoring indicates that the permittee's waste type and disposal methods are in compliance with the permit, then the next concern to be addressed is "Are the LPCs being met in the nearfield (within the site boundaries)?" This question is based on the assumption that disposal operations that comply with the LPCs will be environmentally acceptable.

If the LPCs are met in the nearfield, the next question should be "Are residual concentrations of all pollutants (considered individually and collectively) beyond the site boundaries high enough to cause concern for bioaccumulation and synergistic effects?" However, if disposal operations that comply with existing permit requirements cause the LPCs to be exceeded in the nearfield, it may be appropriate to determine whether existing permit conditions can be revised so that the LPCs are met. For example, the LPCs may become attainable by establishing new disposal rates in the permits. If it is determined that the LPCs can be met by revising the permits, the wastes and disposal operations should continue

to be monitored to be certain that they comply with the new permit requirements. Although Figure 6 depicts a generic decision process and all questions have been answered either Yes or No, it should be noted that monitoring results may indicate that LPCs are inconsistently met in the nearfield. In this situation, the site manager may decide to revise the permits by placing more stringent controls on the disposal process or to proceed with dumping under existing requirements to determine if the rate with which LPCs are exceeded results in undesirable, long-term impacts both within and beyond the site boundaries.

If it is determined that revision of the permit conditions would not allow the LPCs to be met, the need for ocean disposal of the waste compared with other available alternatives should be reevaluated. If this evaluation determines that there are alternatives to ocean disposal that would result in less impact to the environment, then the decision may be made to discontinue ocean disposal in favor of those alternatives. If, however, the results indicate that ocean disposal is more acceptable than available alternatives, then dumping may continue and the question "Is the waste transported beyond the site?" should be addressed.

The need to evaluate whether the waste residuals are leaving the site at elevated levels can be reached through various routes in the decision process, depending on whether the LPCs are being met in the nearfield. However, having reached this point, the questions that follow are the same regardless of the route and involve determination of impacts associated with waste disposal. If it is determined that the wastes are not leaving the site, then the monitoring program is directed to assess nearfield, long-term impacts. If undesirable nearfield, long-term impacts are documented, and relevant null hypotheses thereby refuted, a decision may be made to modify the site designation, or even to de-designate the site. If no

undesirable nearfield impacts can be associated with dumping operations, then dumping may continue for the duration of the permits or the site designation. If the wastes are leaving the disposal site, then the monitoring program results, as prescribed by the relevant monitoring plan tier(s), are evaluated to determine if undesirable farfield, long-term impacts are occurring. If impacts are documented, the site manager may decide to modify the site designation. If impacts are not detected in the farfield, dumping may continue.

It should again be stressed that the decisions that result from this process and the management options that are ultimately exercised must be determined separately for each site and will therefore vary among sites. This section is intended only as a general guidance tool by which site managers can organize and evaluate information obtained through dumpsite monitoring programs.

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OBJECTIVES

- To monitor compliance with permit conditions.
- To provide information necessary to predict the fate and effects of the waste.

NULL HYPOTHESES

WC-A	The quantity and physical/chemical characteristics of the waste are within the permit limits.
DO-A	The waste was discharged within the dumpsite boundaries.
DO-B	The waste was discharged at a rate consistent with the permit.

Figure 1. WASTE CHARACTERISTICS AND DISPOSAL OPERATIONS

OBJECTIVES

- To determine whether nearfield/short-term effects are likely to occur.
- To determine whether farfield/long-term fate and effects monitoring activities are needed.

NULL HYPOTHESES

NF-A The waste plume will follow a trajectory similar (40-200 percent of predicted concentrations) to that predicted by the model used to determine the waste load allocation for the specific site.

NF-B Concentrations of waste constituents are below the applicable water quality criteria within the site boundary four hours after dumping and outside the site boundary at all times.

NF-C Concentrations of floatable materials in the waste are sufficiently low that they will not be transported to shorelines and beaches.

NF-D There are no chemical or physical changes in the water column that affect alkalinity at or immediately adjacent to the site.

NF-E Waste constituents do not settle beneath the seasonal pycnocline within the site boundary.

Figure 2. NEARFIELD FATE

OBJECTIVES

- o To ensure that any short-term effects of waste disposal meet the impact criteria of Section 228 of the oceean dumping regulations.

NULL HYPOTHESES

ST-A There will be no change in planktonic community structure associated with oxygen depletion of the water column attributable to ocean dumping.

ST-B There will be no change in the species composition or productivity of phytoplankton communities in the surface waters of the dumpsite attributable to an increase in the concentration in surface waters of the dumpsite of either primary nutrients or toxic chemicals derived from ocean-dumped wastes.

ST-C The waste plume, following initial dilution in the receiving waters, is not toxic to sensitive species and life stages of marine organisms, as evidenced by acute toxicity and sublethal effects tests performed with site water.

ST-D There is no change in the community structure and diversity of planktonic communities within the dumpsite boundaries that can be attributed to ocean dumping activities.

ST-E Indigenous planktonic and pelagic fauna from the dumpsite do not accumulate contaminants from ocean dumped wastes to concentrations that might be harmful to the animals themselves, or to consumers of the animals, including man.

ST-F Pathogens will not accumulate in the water, sediments, or biota of the dumpsite to concentrations that might pose a health hazard to the indigenous fauna of the site or to people that come in contact with the water or fishery products from the site.

Figure 3. SHORT-TERM EFFECTS

OBJECTIVES

- To determine the direction and areal extent to which the waste is transported beyond the site boundaries and to the ocean floor so that decisions can be made regarding implementation of further monitoring activities.

NULL HYPOTHESES

FF-A The waste concentrations in waters and sediments at specific locations inside and outside the site are 40-200 percent of the concentrations predicted by the model used to determine the waste load allocation for the specific site.

FF-B There is no trend of increasing water column concentrations of wastes in the water column inside or outside the site boundary.

FF-C Waste constituent concentrations do not increase above historical ambient levels within 12 nautical miles of any shoreline, critical area, or marine sanctuary (alternatively, waste constituents do not impinge on any shoreline or marine sanctuary).

FF-D Waste disposal operations do not result in increases in concentrations of contaminants in sediments outside the site boundary.

FF-E Sedimentation rates at and downcurrent of the site are not different from those at upcurrent sites.

Figure 4. FARFIELD FATE

OBJECTIVES

- To ensure that any long-term effects of waste disposal meet the impact criteria of Section 228 of the Ocean Dumping Regulations.

NULL HYPOTHESES

LT-A Species of commercially, recreationally, or intrinsically valuable marine animals from within and adjacent to the site boundary do not accumulate in their tissues any contaminants derived from ocean dumped wastes to concentrations significantly higher than normal background concentrations.

LT-B Residues of contaminants derived from ocean dumped wastes do not accumulate in the edible tissues of any commercial or recreational fishery species collected from within or outside the dumpsite boundaries to concentrations that are in excess of the FDA Action levels or that result in a lifetime risk of cancer of greater than 1/1,000,000, assuming a consumption rate of 6.5 g/day of seafood from the collection site.

LT-C Surface waters from the dumpsite do not show evidence of chronic toxicity or mutagenicity, in comparison to clean seawater from a reference site, based on suitable, sensitive toxicity and mutagenicity tests.

LT-D Benthic community structure does not change over time and in relation to community structure at suitable reference stations at any locations outside the dumpsite boundaries where actual observations or predictions of validated dispersion/fate models indicate that solids from dumped wastes are accumulating.

LT-E Waste dumping operations in the dumpsite do not result in changes in the distribution, migratory patterns, or abundance of any threatened or endangered species of marine mammals or reptiles in or adjacent to the dumpsite.

LT-F The structure of planktonic and pelagic communities within or outside the dumpsite boundaries does not change over time as a result of ocean dumping activities.

Figure 5. LONG-TERM EFFECTS

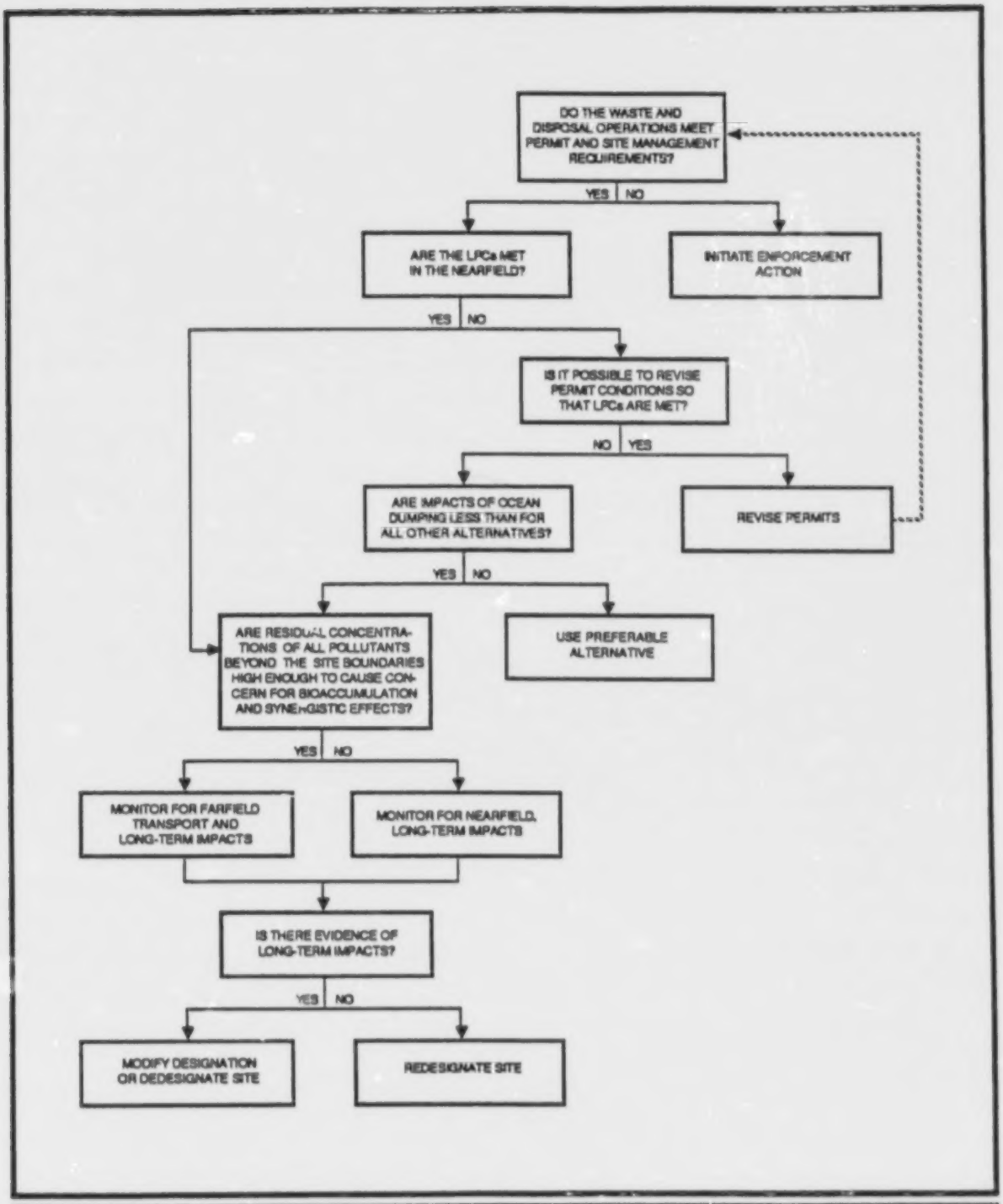

Figure 6. Monitoring results can be used to make monitoring program, permitting, and site designation decisions.

TABLE 1. RELATIVE IMPORTANCE OF EACH SAMPLE HYPOTHESIS BASED ON WASTE AND SITE CHARACTERISTICS

HYPOTHESIS (Descriptor)*	DENSE, SOLID WASTE			LIQUID BUOYANT WASTE		
	Disposal Type			Disposal Type		
	Coastal Contained	Coastal Dispersal	Oceanic Dispersal	Coastal Contained	Coastal Dispersal	Oceanic Dispersal
WC-A Permit	++	++	++	++	++	++
DO-A Boundaries	++	++	++	++	++	++
DO-B Rate	++	++	++	++	++	++
NF-A Plume	-	-	+	++	++	+
NF-B WQC	-	-	-	++	++	+
NF-C Floatables	-	-	-	+	++	-
NF-D Alkalinity	-	-	-	+	+	-
NF-E Pycnocline	++	+	+	-	-	-
ST-A Oxygen	+	+	+	++	++	-
ST-B Phyto	-	-	-	++	+	-
ST-C Toxicity	-	-	-	++	+	+
ST-D Biol Comm	+	+	+	+	+	+
ST-E Bioaccum	++	++	+	++	+	+
ST-F Pathogens	+	+	-	+	+	-
FF-A Model	+	+	+	+	+	+
FF-B Water Qual	-	-	-	++	++	+
FF-C Shoreline	+	+	-	+	++	-
FF-D Sediment	++	+	-	-	-	-
FF-E Sed Rate	++	+	-	-	-	-
LT-A Bioaccum	++	+	++	+	+	+
LT-B Bioaccum	++	+	++	+	+	+
LT-C Chronic	+	+	+	+	+	+
LT-D Benthic	++	+	+	-	-	-
LT-E Endangered	-	-	+	-	+	+
LT-F Pelagic	-	-	-	+	+	+

* Note: Full statement of hypothesis in text

Explanation of Symbols:

- unimportant, little emphasis needed
+ important, some emphasis needed
++ very important

MONITORING RELEASE OF TRACE METALS FROM STABILIZED OIL ASH REEFS

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U.S.A.

Two artificial reefs composed of stabilized oil ash were constructed in about 13 m of water off central Florida in the Atlantic Ocean on April 7, 1987 through the efforts of Florida Power and Light Company and Florida Institute of Technology. In addition to the oil ash reefs, two reefs of concrete block and two benthic control areas containing no artificial material were established. Our research goal is to test the in situ release and biological uptake of selected metals from stabilized oil ash reefs relative to concrete blocks and benthic control areas.

Oil ash contains 10,000-50,000 μg V/g relative to 10-150 μg V/g in typical marine sediments. Concentrations of Cu, Ni and Zn in oil ash also exceed normal levels found in marine sediments. The ash stabilization process was designed to minimize leaching of these potential contaminants into the marine environment.

Prior to construction of the reefs, water samples, sediment and organisms were collected from the study area. In the water column, background dissolved V concentrations were rather uniform at 30 $\pm$ 2 nmol/kg, consistent with coastal and open ocean values for the Atlantic. Observed background dissolved Ni and Cu levels of 2-3 and approximately 1 nmol/kg, respectively, are also consistent with surface seawater concentrations.

Since installation of the reef system, water samples have been collected in crevices and at distances of up to 1 m from the reefs. Dissolved metal concentrations in and around the concrete block reefs have all been at background levels. This is consistent with values of less than 10 μg/g for V, Cu, Ni and Zn in the concrete blocks. In contrast, crevice samples from the oil ash reefs have up to 60 nmol dissolved V/kg seawater, about double background values. At 1 m from the reef, the highest value observed was 37 nmol V/kg. Trends in crevice and near-reef V values are a function of dilution as well as projected decreases over time in V leaching rates from stabilized oil ash. No significant increases above background values were observed for other metals at the oil ash reefs.

Colonization of the reefs has been rapid and a variety of organisms have been collected and analyzed. Some biological uptake of the metals studied has been observed. For example, samples of the bivalve Anomia simplex collected from the oil ash reefs show 2-10 times greater metal levels than observed at the concrete block reefs or under pre-reef background conditions.

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A MODEL FOR THE DISPERSION OF RADIONUCLIDES RELEASED INTO THE DEEP OCEAN

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ABSTRACT

A new numerical model has been designed in France to assess the radiological consequences of low level radioactive waste disposal on the sea bottom of the Atlantic Ocean. It is a box model covering the world ocean with a finer resolution in the North Atlantic and near the Nuclear Energy Agency dumpsite. The main processes involved in the nuclides transfer from the drums to man are modeled or parameterized : the time variation of the nuclides release, the advection and diffusion for the ocean fluid, the adsorption-desorption on particles, the sedimentation and burial of sediments, the transfer through organisms living in the sea (fishes, crustaceans, molluscs, seaweeds, plankton,...). The dose equivalent to critical group members and the collective dose equivalent are assessed for a set of nuclides released from the drums. A sensitivity analysis has been achieved and preliminary results have been obtained with a simplified source term.

INTRODUCTION

The dumping of radioactive waste at sea is a method of disposal which has been carried out in the deep Atlantic for over thirty years. These practices motivated in 1977 a Decision of the Organisation for Economic Co-operation and Development Council which established a Multilateral Consultation and Surveillance Mechanism for Sea Dumping of Radioactive Waste. Under the term of the Mechanism, Nuclear Energy Agency (NEA) is requested to periodically assess the suitability of dumping sites proposed or used by participating countries.

The current dumping site in the deep North East Atlantic is a rectangle bounded by latitudes 45°50'N and 46°10'N and by longitudes 16°00'W and 17°30'W. To increase the scientific data base related to this dumpsite area, and to develop a site specific model of transfers of radionuclides from the dump area to human populations, NEA established

in 1981 an international "Co-ordinated Research and Environmental Surveillance Programme (CRESP)".

CRESP experts defined a modelling framework for the radiological assessment : models of the release from waste forms, benthic boundary layer, sorption-desorption on particles, settling of particles, dispersion in ocean and coastal waters, biological transports and pathways to man are integrated in a composite model to assess the risk associated with the sea dumping practice. A numerical integrated model has been recently developped in France according to this modelling framework. This paper describes the model, and displays the results of a sensitivity analysis on the marine transfer submodel. An assessment of the risk associated with the past dumping of radioactive waste in the NEA site has been calculated with the model. The results are discussed and compared with recommended limits and risks associated with natural radioactivity.

DESCRIPTION OF THE FRENCH/CEA MODEL OF TRANSFER IN THE WORLD MARINE ENVIRONMENT

The model of nuclides transfer is of "compartment" type. The world ocean and the upper layer of the bottom sediment have been divided in 55 boxes. Figure 1 describes the horizontal geometry of the boxes which has been designed to fit the gross features of the North Atlantic Ocean Topography. A number of boxes have been added to model the coastal seas, because some important radiological pathways originate in coastal waters (cf. following sections). The vertical structure of the ocean model (figure 2) is based on the vertical structure of the water masses and the vertical shear of the current. The bottom sediment is modelled by a sediment box under all deep ocean or deep coastal boxes. The depth of these sediment boxes correspond to the bioturbated layer depth (0.1 m). Between the deep ocean and the sediment, a bottom boundary layer (BBL) boxe has been inserted.

The main nuclides transport processes in the marine environment are explicitely taken into account in the model (figure 3). These are the horizontal and vertical diffusion by the ocean turbulence, the sorption-desorption on particles, the settling of particles, the BBL-sediment interactions and the burial in the sediment. All these processes are modelled in both the coastal and ocean waters.

The ocean circulation pattern comes from a mixture of results of a simulation of the North Atlantic general circulation using the Sandia-Harward general circulation model [1] and data based on a general understanding of the world ocean general circulation. There is a lack of data on the water exchange between the ocean and the coastal seas, and no advection has been allowed between ocean and coastal boxes (transports are only modelled by an eddy diffusion).

The range of particle sizes has been divided in two classes : the small particles (< 50 μm) and the large particles (> 50 μm). The small particles are supposed to descend very slowly so that they are advected and dispersed by the ocean water. Morever, a constant equilibrium has been assumed between the small particles and the surrounding water. On the contrary, the large particles are assumed to fall rapidly from the ocean to the bottom so that transfers of nuclides between water and particles occur only at the surface where large particles are formed.

The numerical model consists in 55 linear equations with 55 unknowns (the nuclide concentration in each box). The numerical schemes to resolve this set of equations have been chosen to insure specific numerical properties : the time step may be chosen rather long (1 to 10 years) to allow long term simulation (100 000 years or more) ; the total quantity of each nuclide, summed over all the boxes, plus the quantity removed by radioactive decay or burial, is conserved at each time step, and is equal to the nuclides input.

SENSITIVITY ANALYSIS OF THE TRANSFER MODEL

The objective of this sensitivity analysis is to investigate the importance of model parameters or assumptions in determining the nuclides concentrations in water and sediment. CRESP experts defined a range of possible value for each physical or chemical parameters, and a "best estimate" value. The methodology consists to vary independently the parameters to the lowest or highest possible value, and to compare the results with those obtained with "the best estimate" value of the parameters. Five radionuclides have been considered in this sensitivity analysis (^{14}C, ^{137}Cs, ^{129}I, ^{230}Th and ^{239}Pu) because they span a large range of radioactive half-lives and partition coefficients K_d.

The following potentially important parameters and assumptions have been investigated : porosity, pore water diffusivity, sediment thickness of the bioturbated layer, bioturbation coefficient, settling velocities of the small and large particles, suspended sediment loads in small and large particles, vertical eddy diffusivity and advection by the ocean flow. Many results have been obtained which are summerized in a synthetic manner in the following.

^{230}Th and ^{137}Cs are the most sensitive nuclides among the five nuclides investigated. ^{230}Th (medium half-life and very high K_d) is sensitive to suspended sediment loads and particles settling velocities (concentration variation range of one order of magnitude in the water boxes). ^{137}Cs (short half-life and rather small K_d) is sensitive to suspended sediment loads, vertical ocean mixing coefficients and advective fluxes (concentration variation range of one order of magnitude in far field ocean boxes).

In sediment boxes, the concentration is particularly influenced by the bioturbated layer thickness and the porosity (variation of concentration of one to four orders of magnitude in some boxes for ^{14}C, ^{129}I and ^{239}Pu). The concentration in sediment boxes is also sensitive to the pore water diffusion coefficient (variations of concentrations of one order of magnitude for all the nuclides).

In the water boxes, the concentration is affected by variations of the general circulation pattern : concentration variations of more than one order of magnitude have been observed in the far field boxes for ^{230}Th and ^{137}Cs. Moreover, the concentration in the near field boxes varies quasi-linearly with respect to the vertical eddy diffusivity coefficient until the peak concentration is reached. This coefficient influences also the concentration in the far field boxes for highly sorbed nuclides.

It should also be noted that not only the concentration but also the time of occurence of the peak concentration varies. For example, when the advective fluxes are set to zero, the peak activity in the water boxes bordering the site is reached more than 100 years later than for the base case, and for the sediment boxes, the difference is about 200/300 years.

To conclude this sensitivity analysis, it is worthy to note that the most sensitive transfer processes are the parameterizations of the sediment-water and particles-water

interactions. One should also emphasized that variation ranges of one or two orders of magnitude have been currently observed. This range of variation gives a gross approximation of the uncertainty on the results.

One should finally point out that none of the possible parameter value recommended by CRESP experts have been found to give a maximum of concentration for all the boxes and all the nuclides. This excludes *a posteriori* any radiological conservative approach for this world marine transfer modelling.

RADIOLOGICAL ASSESSMENT OF THE PAST SEA DUMPING OF RADIOACTIVE WASTE IN THE NORTH ATLANTIC OCEAN

Since 1949, radioactive waste disposal operations in the deep North East Atlantic have been carried out at several locations (figure 4). Since 1971 and up to 1982, most of the activity has been dumped at a single site centered at 46°N and 16°45'W. The assessment presented in this section then concerns the consequences of the dumping of radioactive waste for eleven years at this site.

SCOTT *et al.* [2] have developed a model of the waste package which gives the time variation of the release rate out of the drums for the twenty-one most important nuclides. This model takes into account the inventories of dumped radionuclides and parameterizes the drum failure with respect to time. Since UK disposal operations account for a large fraction of the total waste disposed at the NEA site, and because detailed inventory data were not available for waste disposed by other nations at this site, the waste model only considers the UK waste. The results of this waste package model have been specified as input data in the French transfer model for the radiological assessment described here.

The methodology of the radiological risk assessment consists in calculating doses resulting from the dispersion of the nuclides in the sea water and marine sediment. It is useful to consider two different doses : the radiation dose to the most highly exposed individuals in the human population (often referred to as "critical groups") and the total dose to the human population (also called "collective dose" to the world population). An extended series of specific actual or hypothetical pathways have been

defined by the CRESP experts (table 1). The critical group has been defined as the individuals whose consumption rates, inhalation rates or other habits are extreme and exceptionnal in the world population (table 2).

For the collective dose calculations, the five a priori dominant pathways have been considered : direct or indirect consumption of fish, molluscs, crustaceans, seaweed and plankton. Data on the present catches of seafood have been gathered [3]. These catches have been assumed constant in the future, since they are considered as nearly maximum at present, given the stock and the fishing effort.

For two highly mobile radionuclides, namely ^{14}C and ^{3}H, results of other models [4] of global dispersion of these nuclides in the biosphere have been compiled and included in the collective dose calculations.

The concentration factors (defined as the ratio of the concentration of an element in a material or organism to the concentration in the surrounding water) and the dosimetric data (which relate the dose received by body organs to contamination) used in the model have been updated by CRESP experts.

Figure 5 gives the time variation of the maximum dose to critical group individuals. The radionuclides (or the precursor nuclide of a decay chain), pathways and boxes which give the dominant contributions to this dose, are also mentionned on figure 5. The peak annual individual dose is then 2.5 10^{-7} $Sv.y^{-1}$. The dominant contributor are ^{241}Pu and ^{241}Am, via the "mollusc" pathway, these molluscs being catched in the North European continental shelf water (not far from the dumpsite). The peak acti-vities in these boxes, which give the peak individual dose, are still very low : 1.1 10^{-4} $Bq.m^{-3}$ for ^{241}Am and 1.3 $10^{-4}Bq.m^{-3}$ for ^{241}Pu. These levels of activity are likely to be of the same order of magnitude, if not less, as the background activity levels originating in other practices (weapon tests, reprocessing plant discharges...).

It is useful to compare the peak individual dose to the internationationally recognised limits of the International Commission on Radiological Protection [5] of 5.10^{-3} $Sv.y^{-1}$ for the average annual dose to members of the public. Even with an uncertainty of several orders of magnitude, the peak individual dose remains far under the ICRP limit. Also informative is the comparison of the peak individual dose due to

the past deep sea dumping practice with the dose from natural sources which is estimated to about 2.10^{-3} Sv.y^{-1} in areas of normal background activity [6].

The collective dose to the world populations, integrated over 100 000 years (the collective dose remains nearly constant after 10 000 years) is 1.8 10^4 man.Sv. This collective dose is very low indeed compared to the collective dose to the world population from natural sources which is estimated to about 8.10^6 man.Sv *per year*. Moreover, because of the large dispersion of the nuclides in the world ocean, the collective dose due to the deep sea dumping is made up of very small doses to a large number of people.

Table 3 gives the contribution of the dominant nuclides in the collective dose. This dose is dominated by the ^{14}C contribution ; the ^{239}Pu contribution is also significant.

To conclude this radiological assessment of the effects of the deep sea dumping of radioactive waste in the North Atlantic, one should emphasize that the results, although preliminary, clearly show that both the individual doses to critical group members and the collective doses are very small. The expected uncertainty on the results, of a few orders of magnitude, is nevertheless considered as rather large and future modelling effort will be devoted to reduce this range of uncertainty.

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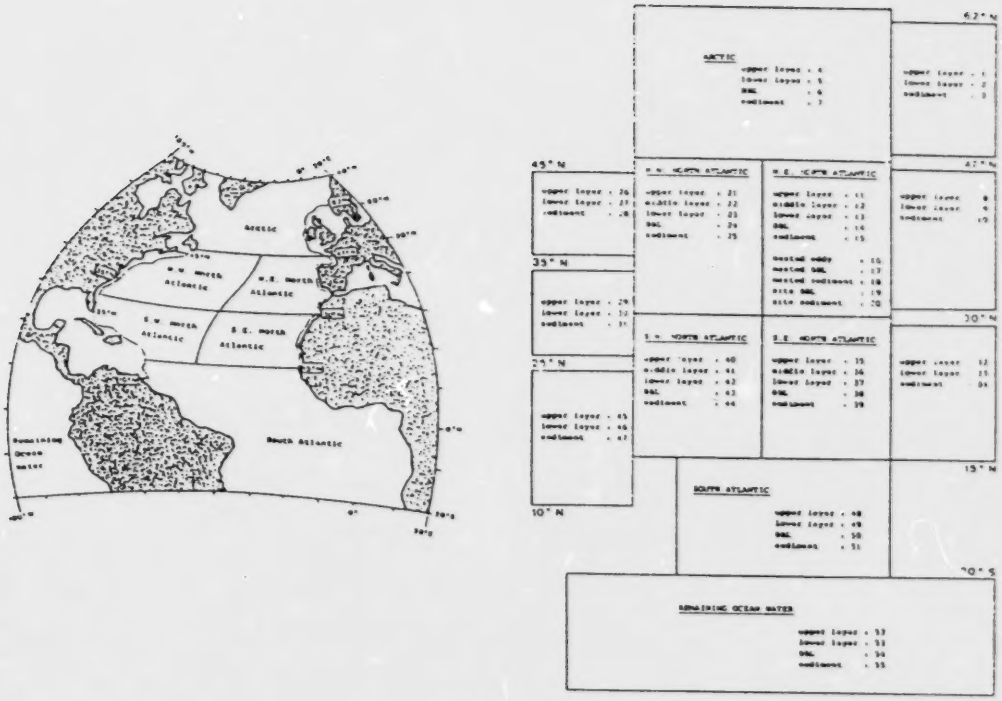

Figure 1

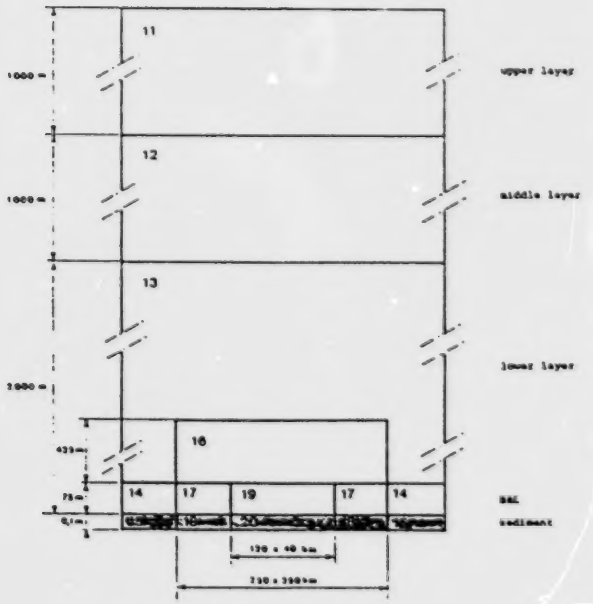

Figure 2

Figure 3

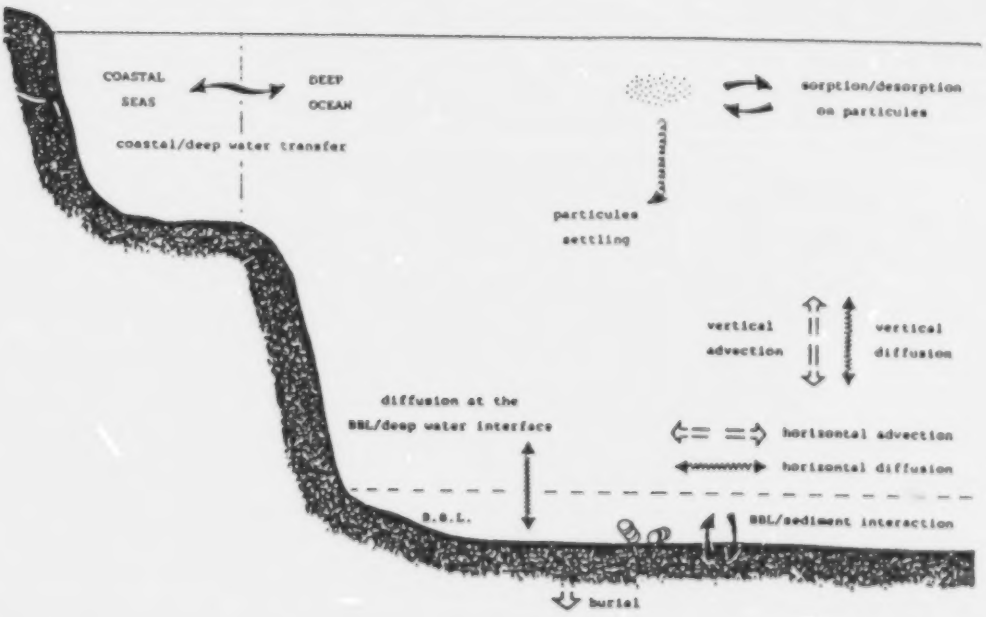

Figure 4

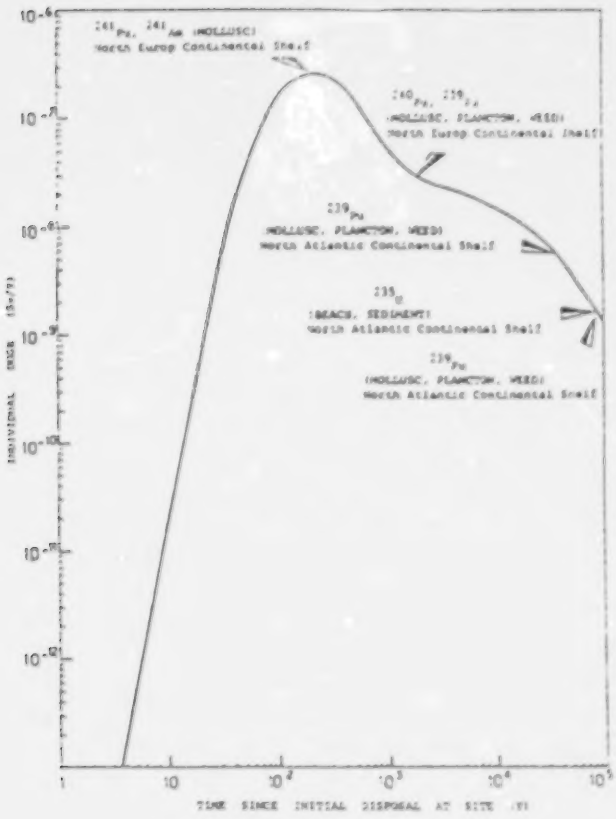

Figure 5

PATHWAY	MODE OF EXPOSURE
ACTUAL PATHWAYS :	
SURFACE FISH CONSUMPTION	INGESTION
MID-DEPTH FISH CONSUMPTION	
CRUSTACEA CONSUMPTION	
MOLLUSC CONSUMPTION	
SEAWEED CONSUMPTION	
SALT CONSUMPTION	
DESALINED WATER CONSUMPTION	
SUSPENDED AIRBORNE SEDIMENTS	INHALATION
MARINE AEROSOLS	
BOATING	EXTERNAL IRRADIATION
SWIMMING	
BEACH SEDIMENT	
HYPOTHETICAL PATHWAYS :	
DEEP FISH CONSUMPTION	INGESTION
PLANKTON CONSUMPTION	

Table 1

PATHWAY	INCORPORATION RATE OR OCCUPANCY FACTOR	
INGESTION RATES :		
FISH CONSUMPTION (S. W)	600 G/D	
CRUSTACEA CONSUMPTION	100 G/D	
MOLLUSC CONSUMPTION	100 G/D	
SEAWEED CONSUMPTION	100 G/D	
SALT CONSUMPTION	2 G/D	
DESALINED WATER CONSUMPTION	2 L/D	
DEEP FISH CONSUMPTION	60 G/D	
PLANKTON CONSUMPTION	2 G/D	
INHALATION RATES :		CONCENTRATIONS :
SUSPENDED AIRBORNE SEDIMENTS	23 M3/D	10 UG/M3
MARINE AEROSOLS	23 M3/D	10 G/M3
OCCUPANCY FACTORS :		
BOATING	5000 H/Y	
SWIMMING	200 H/Y	
BEACH SEDIMENT	2000 H/Y	

Table 2

RADIONUCLIDE	RELATIVE CONTRIBUTION	RADIONUCLIDE	RELATIVE CONTRIBUTION
^{14}C	76.5 %	^{235}U	0.7 %
^{239}Pu	17.1 %	^{242}Pu	0.6 %
^{241}Pu	3.9 %	^{226}Ra	0.3 %
^{240}Pu	1.6 %	^{241}Am	0.3 %

RELATIVE CONTRIBUTION OF THE MAIN RADIONUCLIDES TO THE COLLECTIVE DOSE

Table 3

A THEORETICAL LOOK AT THE IMPORTANCE OF POLLUTANT SCAVENGING AND RETENTION IN COASTAL ENVIRONMENTS

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A model is developed for coastal systems which incorporates descriptions of the sorption properties of chemicals with dilution and scavenging to bedded sediments. The three components of the model calculate the fraction of a chemical on particles, a first-order scavenging rate constant, and the fractions of a chemical which is removed to sediments relative to that which is removed by circulation. The fraction removed to sediments depends on the nature of the circulation in the system so the model calculates the limiting cases of plug-flow and well mixed circulations. In order to run the model, the suspended particle concentration, a scavenging rate for a calibration chemical, and the residence time of water must be available for the system.

The model is calibrated for Narragansett Bay and the New York Bight using scavenging rate data for natural thorium. Chemicals which are subject to scavenging, i.e. those which would be called particle reactive, are different in different systems and may be different seasonally within one system. Many potential pollutants discharged into coastal environments will be primarily trapped in coastal sediments and will not escape to be diluted in the open ocean.

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TRANSFER OF Cu, Zn, Cd, Pb, As, Fe and Mn IN WASTE WATER/SEA WATER MIXED SYSTEM IN COASTAL AREA OF JINZHOU BAY, BOHAI SEA

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ABSTRACTS

In this paper a laboratory simulating experiment on transfer of Cu, Pb, Zn, Cd, As, Fe and Mn in waste water/sea water mixed system in coastal area of Jinzhou Bay, Bohai Sea is described.

After the acid wash waste water from zinc smelter has been drained into sea water the transfer process of heavy metal ions in the mixed system is mainly controlled by adsorption and coprecipitation by $Fe(OH)_3$, as well as adsorption by clay minerals from resuspended sediment. Suspended sediment acts as a nuclear in flocculation. It promotes and accelerates the flocculation rates of Pb, Cu and Zn along with the flocculation of $Fe(OH)_3$,

This laboratory experiment includes the flocculation rates of metals under different waste water/sea water ratios; effects of pH, iron concentration, mixing time and adding suspended sediment on flocculation rates of metals.

Results from the above experiments show that over 90% of Pb, 80% of Cu and 30-40% of Zn in waste water can be Scanvanged by $Fe(OH)_3$ and transferred into marine sediment.

There is no adsorption and coprecipitation of Cd by $Fe(OH)_3$ or suspended clay minerals in the mixed system. The transfer of Cd is mainly controlled by adsorption and complexation of Cd by humic materials with the presence of humic substance in sea water and sediment.

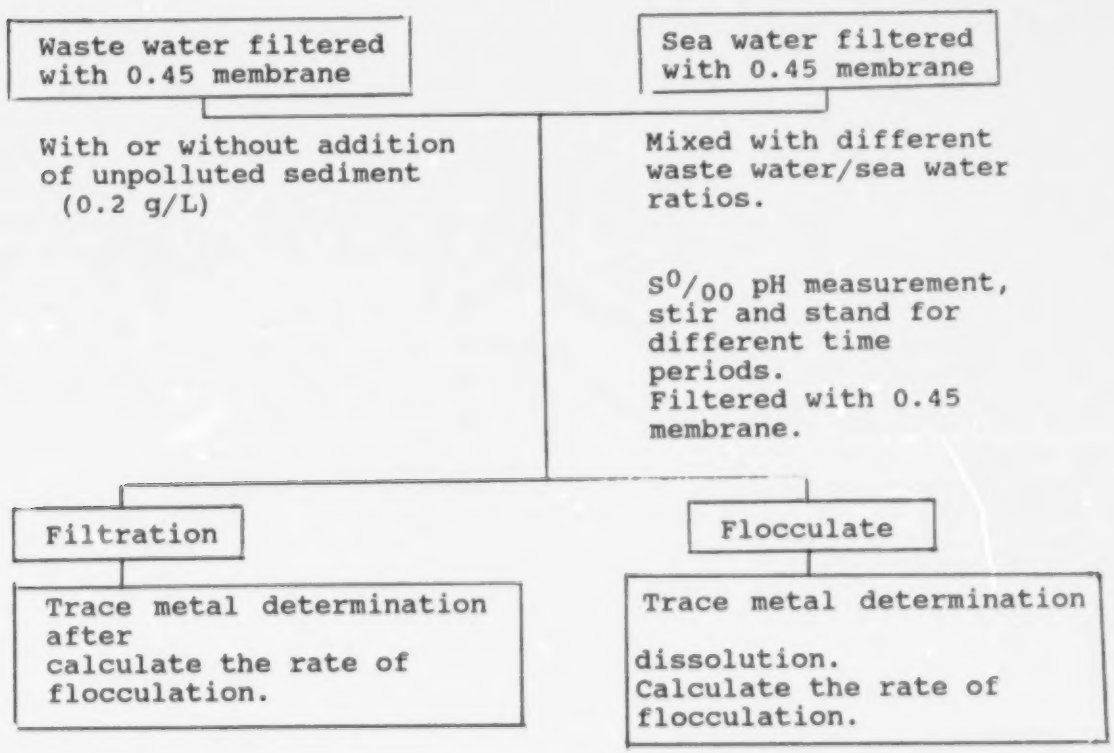
Heavy metal pollution of estuarine and coastal areas results from river and atmospheric input as well as direct drainage of industrial wastes into sea water from coastal industrialized cities. The physico-chemical behaviour of harmful metals in various estuarine and coastal areas is

different with different pollution sources, hydrochemical characteristics and geographic features. Case studies on speciation and transfer of harmful trace metals is not only of practical importance on protection of marine ecosystem in these coastal areas, but also of theoretical meaning of marine environmental geochemistry of trace metals.

EXPERIMENTS

Laboratory flocculation experiment

The flowsheet of flocculation experiment is as follows:



Trace metal determination in flocculates

Cu, Pb, Zn, Cd, Fe, Mn in flocculate were determined by flame AAS after dessolution. As was determined by arsenic hydride AAS.

RESULTS AND DISCUSSION

The sources of heavy metal pollutants are the outfalls of zinc smelter located at the coast of JINZHOU Bay, BOHAI Sea. The map showing the location of the outfalls is shown in figure 1 and the characteristics and metal concentrations in waste water are shown in table 1.

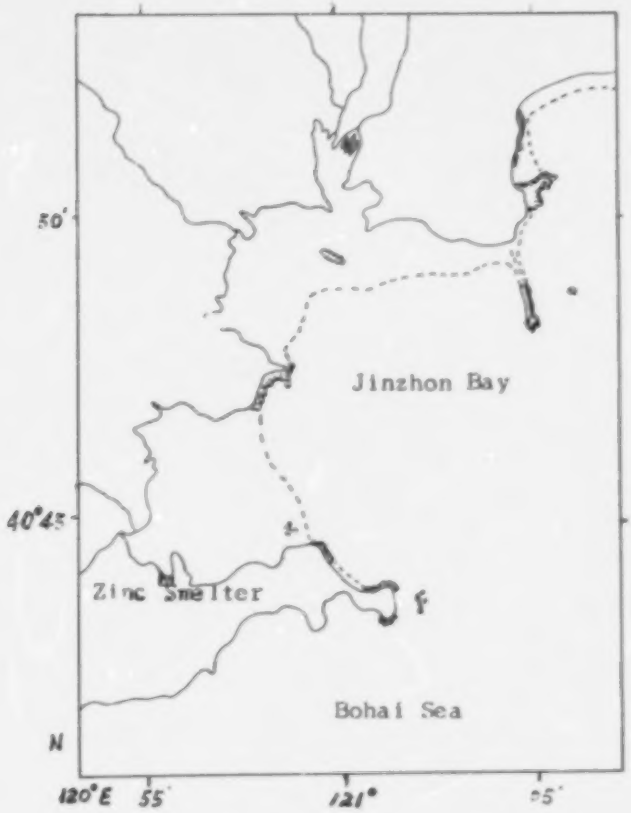


Fig. 1. Map showing the location of zinc smelter at the coast of JINZHOU Bay

Table 1: The characteristics and metal concentrations in waste water (ug/mL)

No. of waste water sample	1	2	3
pH	0.95	1.43	2.18
Element	Concentration ug/ml		
Cu	0.1	3.34	2.66
Zn	61.79	66.75	103.7
Cd	0.15	7.99	12.25
Pb	0	9.05	12.46
Fe	10.2	143	47.34
Mn	8.6	6.2	4.36
As	0.004	2.75	2.08

As we can see from Table 1, all waste water samples are of acidic with very low pH values and contain iron and manganese. In addition, all waste water samples contain certain amount of suspended particles. Obviously, hydrated Fe/Mn oxides will be formed after mixing of waste water with sea water and will act as the best carrier of other trace

metals in transfer from mixed water into sediment (Lee, 1975; Forstner, 1983; Theobold, 1963). And the suspended particles originally exist in waste water or resuspended from bottom sediment will act as a nuclear in flocculation.

1. Flocculation rates of trace metals under various waste water/sea water ratios.

Filtered waste water samples were mixed with filtered sea water in different ratios, stir and stand for about 30 hrs. The flocculates were then filtered and dissolved. Concentrations of Fe, Mn, Cu, Zn, Cd, Pb and As in dissolved flocculates, original waste water and filter membrane were determined simultaneously. Results are shown in Table 2.

Table 2: Flocculation rate of metals under different waste water/sea water ratios

No.	1	2	3	4	5	6	7	8	9	10
waste water/ sea water	1/1	1/3	1/5	1/7	1/9	1/19	1/29	1/39	1/49	1/99
Total volume (ml)	200	200	200	200	200	200	200	200	200	200
PH (after filtration)	1.61	1.91	2.11	2.26	2.39	2.96	4.03	7.25	7.60	7.90
S o/oo	28.639	30.162	30.828	30.962	31.082	31.462	31.218	31.289	31.345	31.453
Element	Flocculation rate (o/o)									
Fe	0	0	0	0	0	0	27.8	86.1	91.5	107.6
Mn	0	0	0	0	0	0	0	0	0	0
Cu	0	0	0	0	0	0	0	67.03	70.1	82.4
Pb	0	0	0	0	0	0	3.3	81.1	88.8	(82.04)
Zn	0	0	0	0	0	0	3.4	24.5	37.3	40.8
Cd	0	0	0	0	0	0	0	0	0	0
As	0	0	0	0	0	0	72.04	103	93.14	107

Results from Table 2 show that there is no flocculation of Mn and Cd under given conditions. In case of Fe flocculation begins from pH 4 and sea water/waste water ratio = 1/29, and the flocculation rate increases with the increasing of pH and sea water/waste water ratio. In case of Pb, Zn and As flocculations begin from pH 4 and Cu from pH 7. The order of flocculation rates is As > Fe> Pb> Cu> Zn. Results indicate that the predominate factor of controlling flocculation is pH value (Balistrieri,1982).

2. Effect of PH on flocculation of metals

Results of flocculation rates of metals under definite waste water/sea water ratio (1/49) and different pH values are shown in Tables 3, 4 and Fig. 2.

Results from Tables 3 and 4 show that the flocculation pH of Fe is 4 and Cu, Zn, Pb are 5. And the flocculation pH of Mn is higher than 8.4. Results also show that formation of $Fe(OH)_3$ greatly promotes the flocculation of Pb,Cu and Zn. The flocculation pH of Pb Cu and Zn are lower than the precipitation PH of Pb, Cu, Zn hydroxides. The order of flocculation rates are also Fe > Pb > Cu > Zn. The role of manganese is in agreement with previous work (Stumm,1960). Hydrated Mn oxide can only be formed under high pH value with low reaction rate. Therefore, it is iron rather than manganese which plays an important role in transfer of As, Pb, Cu, Zn from water phase into solid phase in a waste water/sea water mixed system.

Table 3: Effect of pH on the flocculation of metals. (waste water/sea water = 1/49).

No	1	2	3	4	5	6	7
PH	3.04	3.95	5.05	6.0	7.10	8.04	8.87
S°/₀₀	31.701	31.441	31.412	31.419	31.387	31.364	31.328
Element	Flocculation rate (%)						
Cu	0	0	25	74.9	86	95.6	95.6
Zn	0	0	(1.3)	16.4	45.5	76.9	86.2
Cd	0	0	0	0	0	0	4.0
Pb	0	0	44.6	96.0	96.0	100	98.2
Fe	0	15	104	107	107	107	107
Mn	0	0	0	0	0	0	62.1
As	4.9	66.2	96.8	98.0	98.0	96.8	91.9

Table 4. Effect of pH on flocculation of Mn.

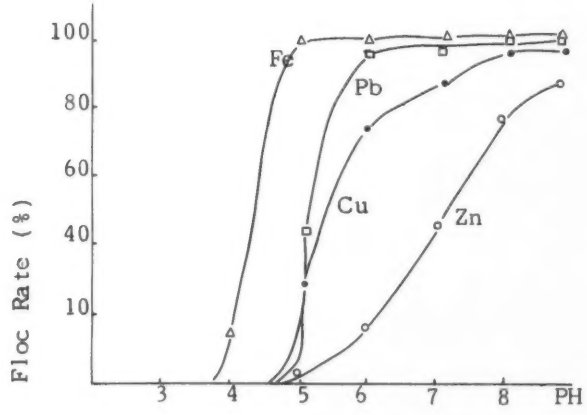


Fig.2 Effect of Ph on flocculation of metals.

3. Effect of iron concentration on flocculation rate of metals

Experiment results of flocculation rates of metals under definite waste water/sea water ratio and different Fe concentrations are shown in Table 5.

Table 5: Effect of Fe concentration on flocculation rates of metals (waste water/sea water = 1/49 pH 8.2)

No.	1	2	3	4	5
Total volume(ml)	200	200	200	200	200
Total amount of Fe (ug)	0	200	516.4	1000	1600
Element	Flocculation rate (%)				
Cu	21.06	84.03	91.9	97.2	98.1
Zn		31.99	93.4	92.9	97.1
Cd		0	0	0	11.7
Pb		91.4	92	97.6	100
Fe		103.8	97.6	98	96.5
Mn		0	0	0	0
As			101	98.0	98.0

The flocculation rates of metals increase with increasing Fe concentration. Cu and Pb in mixed water system were almost entirely flocculated when Fe concentration is 1 ppm. In mixing process of waste water and sea water increase of Fe concentration is helpful to remove harmful metals from water phase and transfer to bottom sediment.

4. Effect of mixing time on flocculation rates of metals

Results of flocculation rates of metals under different mixing time are shown in Fig. 3, 4. From Fig. 3, 4 we can see that: (1) the order of flocculation rates is Fe Pb Cu Zn; (2) there is no transfer of Cd from water phase to sediment; (3) flocculation of Fe Pb begins at 10 minutes and reach to an equilibrium at 8 hrs.; (4) flocculation rate of Zn increases very slightly in the whole mixing period.

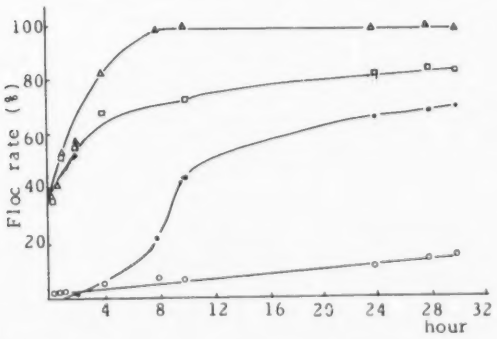


Fig.3 Effect of mixing time on flocculation rates of metals (waste water/sea water = 1/49, PH = 7.6)

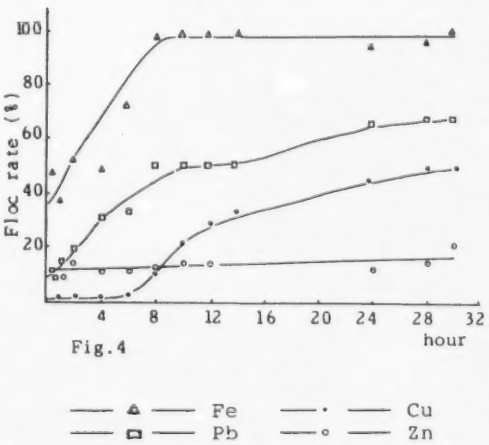


Fig. 4 Effect of mixing time on flocculation rates of metals (water water/sea water = 1/99, pH 7.9)

5. Effect of adding suspended sediment on flocculation rates of metals

Results of adding suspended sediment on flocculation rates of metals are shown in Fig. 5. Results from Fig. 5 show that: (1) the flocculation rates of metals increase greatly in comparison with those without adding suspended

sediment at same mixing time (Aston, 1973); (2) 90% of Pb is transferred from mixed water into sediment at mixing time of 10 minutes and almost entirely removed within 2 hours; (3) similar effects can be seen in the case of Cu and Zn; (4) the probable mechanisms of adding suspended sediment are nucleation and adsorption, the absorption ability of Pb, Cu and Zn by suspended sediment is in good agreement with the empirical sequence for the affinity of heavy metals toward clay minerals established by Mitchell (Mitchell, 1964): Pb > Ni > Cu > Zn.

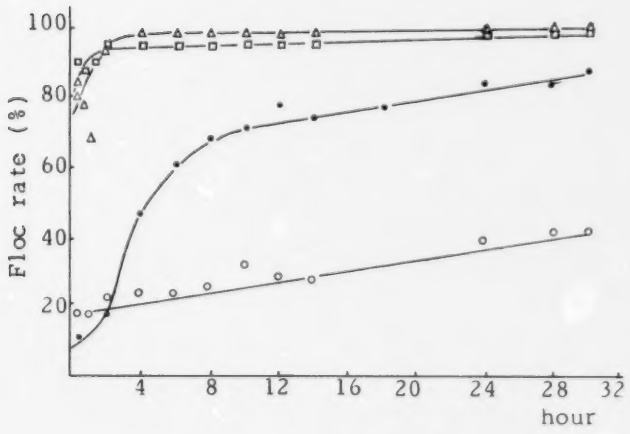


Fig.5. Effect of adding sus-sediment on flocculation rates of metals. (waste water/sea water = 1/99 PH = 7.9)

——— ^ ——— Fe

——— ▫ ——— Pb

——— · ——— Cu

——— ◦ ——— Zn

6. Transfer mechanism of cadmium

Results from Table 2 show that there is no coprecipitation of Cd by $Fe(OH)_3$ or adsorption by suspended sediment (clay minerals) in waste water/sea water mixed system under pH 8.

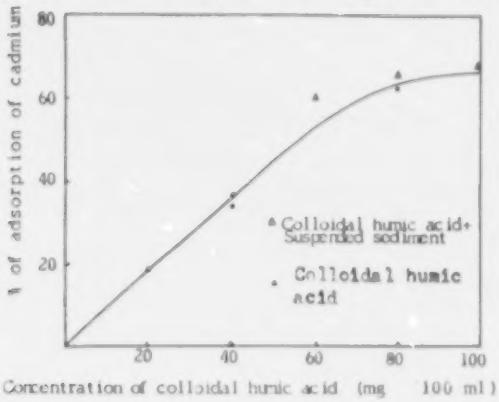


Fig.6 Adsorption of cadmium by colloidal humic acid and suspended sediment

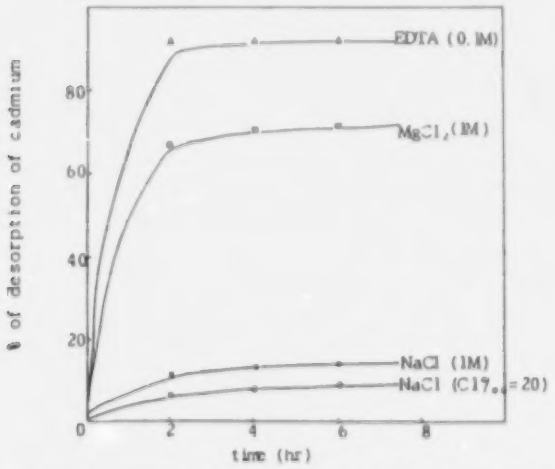


Fig.7 % of desorption of humic substance combined cadmium vs time

Results from Fig. 6 show that cadmium can be strongly adsorbed by colloidal humic acid and the percentage of adsorption of Cd increases with the increasing of concentration of humic acid. There is almost no additional adsorption of Cd by adding suspended sediment. It explains

that transfer of cadmium is mainly controlled by adsorption and complexation of Cd by humic substances existed in water phase and sediment.

Fig. 7 indicates that cadmium can be transferred from humic substance combined state in sediment into sea water by exchanging with Nacl, $MgCl_2$ or EDTA solutions, i.e. Cd can be existed either in sediment or in sea water. It depends on the relative concentrations of major ions and organic complexation ligands in sea water and the humic substances in sediment. In case of high metal concentrations in sea water and low humic substances in sediment Cd mostly exists in water phase. And it mainly exists in sediment while the humic substance concentration is high in sediment.

CONCLUSIONS

1. After the acid wash waste water has been drained into sea water of JINZHOU Bay, BOHAI Sea, the transfer of the harmful heavy metal ions in the waste water/sea water mixed system into coastal marine environment is controlled mainly by adsorption and coprecipitation by hydrated Fe oxide as well as adsorption by clay minerals from resuspended sediment. Suspended sediment acts as a nuclear in flocculation. It promotes and accelerates the flocculation rates of Pb, Cu and Zn along with the flocculation of $Fe(OH)_3$.

2. Over 90% of Pb, 80% of Cu and 30 - 40% of Zn in waste water can be scavenged by $Fe(OH)_3$ and transferred into marine sediment.

3. There is no adsorption and coprecipitation of Cd by $Fe(OH)_3$ or suspended clay minerals in waste water/seawater mixed system. It is in good agreement with author's previous work (Qian, 1983). The transfer of Cd is mainly controlled by adsorption and complexation of Cd by humic material with the humic substances in sea water and sediments. Therefore, due to the high concentration of metals and low level of organic substances in coastal sea water adjacent the outfalls of Zinc smelter, Cadmium is the most harmful and hazardous metal in coastal sea water of JINZHOU Bay.

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LEAD DISPERSION GRADIENTS IN SAN FRANCISCO BAY

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ABSTRACT

The dispersion of contaminant leads in San Francisco Bay is evidenced by lead concentration gradients in both subtidal sediments and intertidal organisms. Lead concentrations of sediments decrease from 220 ug/g in contaminated sediments within the central bay to 1 ug/g in adjacent coastal sediments. Similarly, lead concentrations of mussels (Mytilus californianus) decrease from 8 ug/g in mussels transplanted to the south bay to 0.2 ug/g in native mussels in coastal waters. Those gradients are substantiated by lead isotopic composition gradients in sea water, which demonstrate that there is a relatively limited oceanic flux of U.S. industrial leads from the estuarine system. Concentrations of lead within the bay have also remained relatively constant while atmospheric inputs of industrial leads have decreased by over an order of magnitude during the past decade. Therefore, the estuary appears to have been the principal sink for industrial leads discharged to it during the past 200 years. It also appears that the estuary will remain a principal source of contaminant leads for estuarine organisms for a protracted period.

INTRODUCTION

An analysis of problems relating to the San Francisco Bay-Delta (Figure 1) was recently completed by the Aquatic Habitat Institute for the Environmental Protection Agency (Gunther et al.,1987). It concluded that there was not a consolidated data base to accurately establish (a) comparative degrees of pollution of different areas within the estuarine system or (b) changes in such contamination patterns with time. It was also concluded that the worth of the existing data base on contaminants in San Francisco Bay was limited by the insufficient weight accorded to quality assurance and control issues in the past.

Those limitations are especially valid for lead. There is no comprehensive data base on lead contamination in San Francisco Bay and problems of inaccurate lead measurements are now widely recognized. The latter problem was specifically noted by Bruland (1983) in his review on trace elements in sea water:

> "The decrease of three orders of magnitude in the accepted concentration of lead in sea-water over the last four decades is not a real effect but an artifact of successive improvements in reduction and control of the level of contamination introduced during sampling, storage and analysis (Schaule and Patterson, 1981)".

The problems of inaccurate lead measurements have not been limited to sea water analyses, as demonstrated by Settle and Patterson (1981) in their intercalibration of lead analyses in fish tissue. It has also been demonstrated by analyses of other matrices by Patterson and his associates.

In spite of those limitations, geographic gradients of lead concentrations in the San Francisco Bay system are apparent. These indicate spatial gradients in the dispersion of industrial leads within the system, which are comparable with those of natural ^{210}Pb (22.3 year half-life) in other estuarine systems (Turekian, 1977). This is consistent with oceanic data, which exhibit parallels in the distributions of stable lead and radiogenic lead in surface waters (Schaule and Patterson, 1981; Settle et al.,1981; Flegal and Patterson, 1983; Boyle et al.,1986).. The latter covariance is due to the current predominance of aeolian inputs of both industrial and radiogenic leads to those waters (Flegal and Patterson, 1983).

There does not appear to be a corollary temporal decrease in lead concentrations within San Francisco Bay associated with

decreases in atmospheric emissions of industrial lead aerosols within the past decade. This contrasts with the decreases reported in other estuarine and oceanic systems, which were attributed to to the systematic reduction in leaded gasoline emissions within the United States that was initiated in 1973. Those include a 15% decrease in lead concentrations in the upper thermocline of the North Atlantic between 1979 and 1984 (Boyle et al., 1986) and a 40% decrease in industrial lead transported to the ocean by the Mississippi over the past decade (Trefry et al.,1986).

The apparent absence of a temporal decrease in lead concentrations within San Francisco Bay directly contrasts with the pronounced decreases of leaded gasoline emissions within the bay area, which meet or exceed those observed on the national level. It indicates that sediments within the bay have been the principal sink for contaminant leads that have been directly discharged to the bay for the past 200 years (Nichols et al.,1986). It also indicates that those sediments have now become a principal source of contaminant lead for organisms within the bay and will remain so for a protracted period.

MATERIALS AND METHODS

Much of the data presented in this report has been derived from other studies. Atmospheric lead concentrations were obtained from the California Air Quality Data reports of the Bay Area Air Quality Management District. Sediment lead concentrations were obtained from the recent summary by Long (1987), who collated from data from numerous studies for the U.S. Army Corps of

Engineers, U.S. Geological Survey and National Oceanographic and Atmospheric Administration. Mussel lead concentrations were obtained from the national (Goldberg et al., 1983) and state mussel watch (Smith et al.,1986; Stephenson et al., 1985) programs. Sea water analyses were obtained from studies by Gordon (1980).

Other data,which have not been reported previously, were obtained with established procedures. Surface water samples were collected from a raft and stored with the procedures described in Flegal and Patterson (1983). Filtered (0.4 um) lead concentrations of samples from the bay were measured by graphite furnace atomic absorption spectrophotometry following an organic (APDC/DDDC) extraction using the procedures described in Gordon (1980) and Martin et al.(1986). Lead concentrations of samples from coastal waters were measured by isotope dilution thermal ionization mass spectrometry using the procedures described in Flegal and Patterson (1983). Lead concentrations of mussels and sediments were measured by graphite furnace atomic absorption spectrophotometry using the procedures developed for the National Mussel Watch (Goldberg et al., 1983). Limits of detection, sampling and analytical blanks, precision, and accuracy based on concurrent analyses of reference materials from the National Bureau of Standards of all analyses have been comparable to those reported in those references.

ATMOSPHERIC INPUTS OF INDUSTRIAL LEADS TO THE BAY

Direct and indirect inputs of industrial lead aerosols are a principal source of contaminant lead in the bay. Direct

atmospheric loadings are now estimated to be between 6 and 20 tonnes per year and to be from 1% to 17% of the total mass loadings to the system (Gunther et al.,1987). Atmospheric inputs may also account for most of the estimated loadings (Gunther et al., 1987) from urban runoff (30 to 250 tonnes per year), nonurban runoff (25 to 350 tonnes per year) and riverine inputs (30 to 75 tonnes per year). Most industrial lead emissions are deposited within the bay's drainage basin due to the relatively short (10 day) atmospheric residence time of lead aerosols, as indicated by the predominance of atmospheric inputs of Pb^{210} in Long Island Sound (Benninger et al., 1975). Therefore, the sum of these atmospheric inputs of industrial leads to the bay may approach 116 to 685 tonnes per year and represent 53% to 71% of the total lead input to the system.

This estimate of atmospheric inputs to the bay is consistent with measurements that have demonstrated the predominance of atmospheric lead inputs (>95% of all inputs) in adjacent oceanic surface waters of the North Pacific (Flegal and Patterson, 1983). It is also comparable with estimates of the predominance (>40%) of atmospheric inputs to the Great Lakes (Hodson et al., 1984; Schmidt and Andren, 1984). The latter comparison with the Great Lakes is appropriate since both systems are similar in their relatively urban surroundings and land-locked configurations.

As previously noted, there has been a marked decrease in the emissions of industrial lead aerosols in the San Francisco Bay area, which meet or exceed those on a national level. Production and consumption data indicate that lead usage in gasolines within the United States decreased over 85%, from 443.5 metric tonnes

per day in 1976 to 68.5 metric tonnes per day in 1985. The same data indicate that decreases within California (91.5%) and the San Francisco Bay area (91.2%) have been even greater (Chestnut et al., 1987).

The decreases in the usage of leaded gasolines is reflected in the decreases (2.4 ug/m^3) to 0.1 ug/m^3) in atmospheric lead concentrations in urban areas surrounding the bay (Figure 2). The order of magnitude decreases in the atmospheric lead concentrations indicate that the total loadings of the San Francisco Bay-Delta system have decreased by at least one order of magnitude during the past decade due to reductions in emissions of industrial lead aerosols, based on the preceding estimates of Gunther et al. (1987).

Geographic differences in atmospheric lead concentrations within the bay (Figure 2) evidence both spatial differences in emissions and the relatively short atmospheric residence time of lead. Spatial differences between urban (San Francisco and San Jose) and relatively rural (Pittsburg) areas have become less significant since the controls of leaded aerosols were initiated. This has minimized geographic gradients in direct inputs of lead aerosols to the bay. The relatively short (10 days) atmospheric residence time of lead aerosols (Turekian et al., 1977) contributes to that geographic gradient, and indicates that most of the lead is deposited within the San Francisco Bay-Delta drainage basin. This accounts for the predominance of atmospheric lead in runoff and riverine inputs to the bay, which is similar to the predominance of atmospheric ^{210}Pb inputs to Long Island Sound (Turekian, 1977).

LEAD CONCENTRATIONS IN SEDIMENTS

Temporal gradients

In contrast to the marked decrease in atmospheric lead concentrations in the San Francisco Bay area, there has been no comparable decrease in the lead concentrations of surface sediments within the bay. This was reported in a recent summary (Long, 1987) of over 1,200 measurements of lead concentrations in surficial sediments of San Francisco Bay between 1970 and 1987 (Figure 3). During that eighteen year period, lead concentrations ($x \pm s$) of the bay's surficial sediments have remained relatively constant (50 ± 125 ug/g). These data are surprisingly consistent, despite the qualifications and the quality control limitations noted by Gunther et al. (1977). Most (87%) of the concentrations are between 10 and 100 ug/g, and relatively small percentages are less than 10 ug/g (8%) or greater than 100 ug/g (5%).

The relatively small temporal differences in sediment lead concentrations appear to be primarily a consequence of different annual sampling regimes, rather than annual differences in lead concentrations of sediments within the bay (Long, 1987). For example, the highest annual (1979) mean concentration (79 ug/g) was derived from sediments collected from relatively contaminated peripheral areas associated with sewage outfalls and the lowest annual (1984) mean concentration (17 ug/g) was derived from sediments collected from relatively uncontaminated basin sediments. The relatively constant lead concentrations of the bay sediments is also consistent with the relatively constant lead concentrations (0.76 ± 0.50 ug/g) of adjacent coastal sediments at four locations over a four year (1982 to 1985) period (Flegal,

unpublished data).

Geographic gradients

Comparisons of the preceding bay and coastal sediment data reveal a pronounced geographic gradient in the lead concentrations of sediments within the bay and in the adjacent coastal zone (Figure 3). Average lead concentrations are highest (up to 10,000 ug/g) in peripheral margins of the bay (71 ug/g), intermediate in the central basins of the bay (29 ug/g) and lowest in the adjacent coastal zone (0.8 ug/g). These decreases in lead concentrations of sediments are nearly an order of magnitude between peripheral and basin areas within the bay, and they are another order of magnitude between basins within the bay and the coastal zone.

The geographic distribution of lead concentrations in the sediments evidences a rapid sequestering of labile lead by fine grained sediments, which are deposited along the periphery of the bay. Lead concentrations in those sediments are highest because of their relatively high surface area, as well as their proximity to near-shore outfalls and surface runoff. The efficiency of that scavenging process apparently accounts for the relative absence of lead contamination in adjacent coastal sediments, where lead concentrations have remained near background levels.

The geographic gradient of lead concentrations in the sediments is consistent with Turekian's (1977) description of the flux of ^{210}Pb to the ocean, which is partially based on analyses of the ^{210}Pb budget in Long Island Sound (Benninger et al., 1975; Benninger, 1976; Lewis, 1976). As with industrial lead in San

Francisco Bay, the predominant source of the radioisotope is atmospheric fallout. Essentially all of the atmospheric input of ^{210}Pb is retained by the soils, with a mean residence time (relative to erosion) of several thousand years. The small percentage of ^{210}Pb which is solubilized in fresh water is further sequestered by particles during stream transport. Soluble humic ^{210}Pb colloids, which are not scavenged in fresh water, are flocculated in the estuarine zone at rates greatly exceeding rates of desorption. Additionally, metals mobilized by anaerobic diagenesis within an estuary tend to be rapidly returned to the sediments by oxidation and precipitation within the upper sediment layer and within the water column. In summary, the estuarine ^{210}Pb cycle is dominated by the release and deposition of the nuclide "virtually entirely within the system" (Turekian, 1977).

LEAD CONCENTRATIONS IN MUSSELS

Temporal gradients

Lead concentrations in mussels also appear to have remained relatively constant over the past decade. This is most evident in the national and state mussel watch data for coastal mussels from Bodega Head (38° 18' 42" N, 123° 04' 07" W), which have been monitored with standardized procedures since 1976 (Figure 4). Concentrations have also remained relatively constant in mussels from Trinidad Head (41° 03' 30" N, 124° 09' 00" W), which is also north of the mouth of San Francisco Bay. Contrasting temporal differences of lead concentrations within the bay, are considered to be a consequence of differences in the suspended particulate

load during the sampling period, which influences the amount of particulate lead ingested by the organisms (Smith et al.,1986), rather than differences in the amounts of lead deposited in the bay.

Geographic gradients

Lead concentration gradients in the mussels are comparable to those of the sediments (Figure 5). The highest concentrations (14 ug/g) occur in mussels adjacent to contaminated sediments in peripheral margins of the bay. The average concentrations of transplanted mussels within the bay (2.9 ug/g) are an order of magnitude greater than those of coastal mussels (0.3 ug/g) at the mouth of the bay (Point San Pedro), and average concentrations of mussels further down the coast are even lower (0.2 ug/g).

The gradient in lead concentrations of the mussels appears to be due to the relative amounts of contaminant lead in suspended particulates ingested by the organisms, rather than concentration gradients in dissolved lead which is bioaccumulated by the organisms. It was recently determined that 92% of the lead in contaminated mussels in Monterey Bay was incorporated in their digestive tracts (Flegal et al., 1987).

LEAD CONCENTRATIONS IN SAN FRANCISCO BAY WATERS

Lead concentrations in San Francisco Bay waters and adjacent coastal waters have remained relatively constant during the past decade, since they were first accurately measured in 1979 (Gordon, 1980). Lead concentrations ($x \pm s$) in central bay waters were 33 ± 14 ng/kg in 1979 (Gordon, 1980), and they were 31 ± 5 ng/g in 1985 (Flegal and Niemeyer, 1987). Lead

concentrations in coastal waters ranged from 7 to 23 ng/kg in 1979-1980 (Gordon, 1980) and from 11 to 25 ng/kg in 1986 (Flegal and Niemeyer, 1987).

Isotopic compositions in the coastal waters indicated that the oceanic input of industrial leads from San Francisco Bay was relatively insignificant (Flegal and Niemeyer, 1987). The $^{206}Pb/^{207}Pb$ ratios of adjacent coastal waters ranged from 1.17 to 1.19. These ratios are characteristic of northeast Pacific surface waters (Flegal et al., 1986; Flegal and Stukas, 1987), which are derived from aeolian inputs of industrial aerosols from Asia ($^{206}Pb/^{207}Pb$ = 1.16) and the United States ($^{206}Pb/^{207}Pb$ = 1.22). There was no apparent increase in United States industrial leads in waters near the mouth of San Francisco Bay.

SUMMARY

The principal sink for contaminant leads introduced to the San Francisco Bay-Delta appears to be the sediments within the bay. This is consistent with budget analyses of the ^{210}Pb cycle in Long Island Sound, which demonstrated that only a small percentage of the ^{210}Pb entering the sound was discharged to the ocean. These natural radioisotope measurements were utilized by Turekian (1977) to predict the fate of metals in the sea, since contemporary trace element analyses of sea water were qualified by problems of contamination.

While those problems of contamination have been resolved by the adaptation of ultra-clean techniques (Bruland, 1983), they have qualified most existing data on lead concentrations in San Francisco Bay (Gunther et al., 1987). The temporal variations and

geographic gradients in lead concentrations of sediments and mussels within the bay system that are summarized in this review are limited by that qualification. The sea water analyses are additionally qualified by their small number.

Despite those qualifications, the data exhibit a remarkable consistency. Atmospheric lead concentrations have systematically decreased by an order of magnitude since the restrictions were placed on the utilization of leaded gasolines in 1973. In contrast, lead concentrations in sediments, mussels and sea water appear to have remained relatively constant. Associated with that temporal constancy are pronounced geographic concentration gradients. These evidence the rapid sequestering of lead by particulates within San Francisco Bay. This is consistent with mass balance calculations of the ^{210}Pb cycle in Long Island Sound. These measurements indicate that the sediments of San Francisco Bay have been the principal sink for contaminant leads introduced to the estuary during the past 200 years. The high concentrations of lead accumulated by those sediments has also made them a principal source of lead contamination to organisms within the bay.

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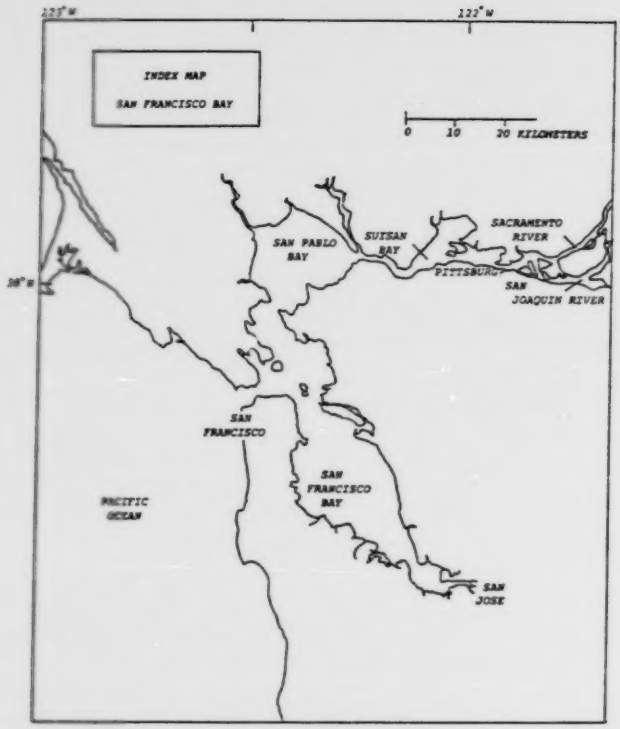

Figure 1. San Francisco Bay and Delta.

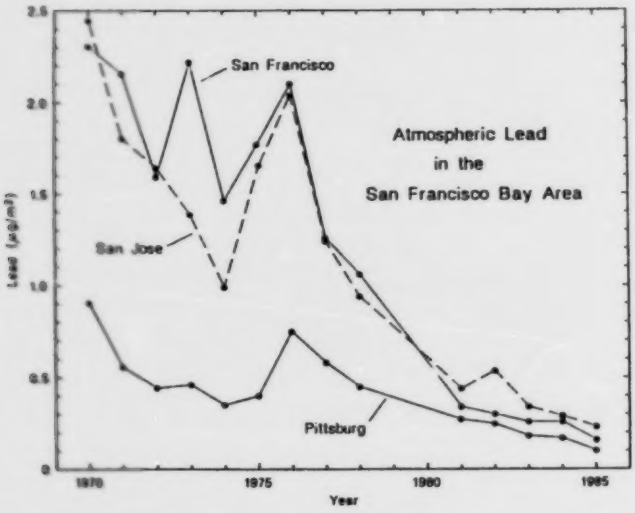

Figure 2. Temporal (1970 to 1985) decreases in atmospheric lead concentrations at sites within the San Francisco Bay area. Data provided by J. Sandberg (San Francisco Air Quality Management District) and the California Air Resources Board.

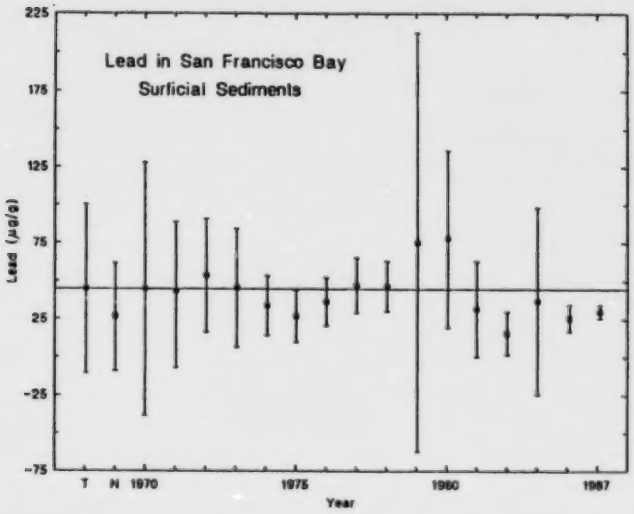

Figure 3. Temporal (1970 to 1985) variations in lead concentrations of sediments in San Francisco Bay (from Long, 1987).

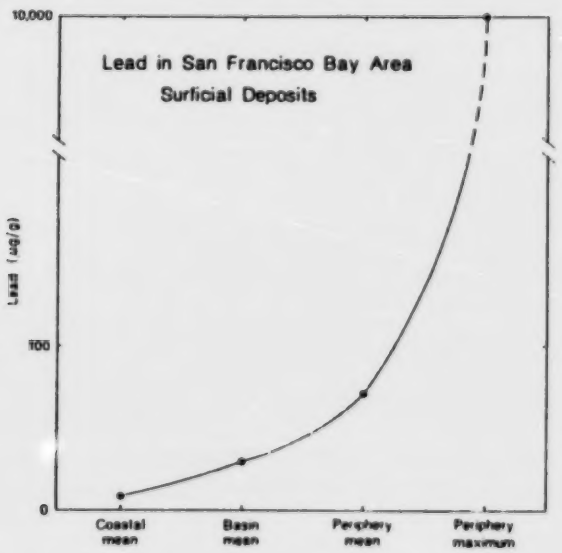

Figure 4. Geographic gradient in lead concentrations of surficial sediments in San Francisco Bay.

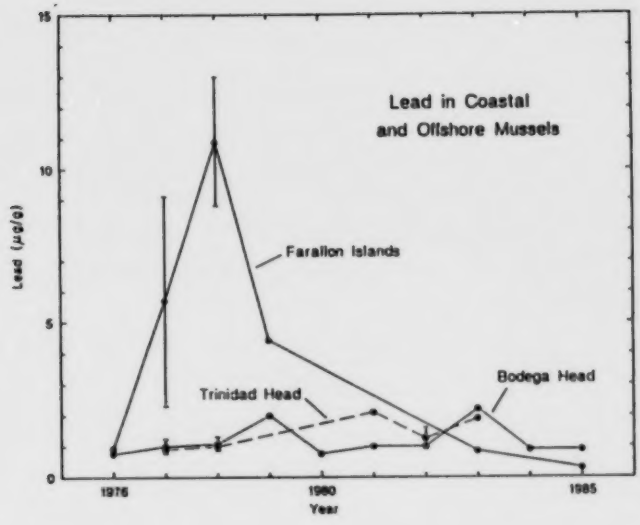

Figure 5. Temporal (1976 to 1985) variations in lead concentrations of coastal and offshore mussels in central California.

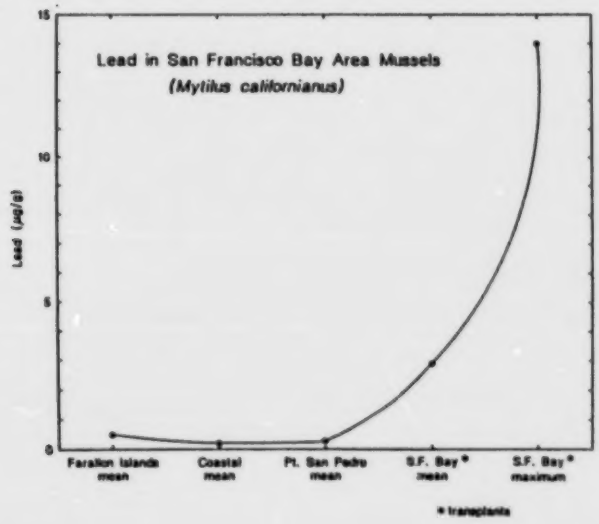

Figure 6. Geographic gradient in lead concentrations of mussels in San Francisco Bay.

Sediment and Biological Conditions in Reference Areas off Southern California, U.S.A.

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SUMMARY

The purpose of the Reference Site Survey was to provide information on environmental conditions from the least contaminated areas on the southern California mainland shelf, and to evaluate changes in conditions at these sites from those reported in the 60-Meter Control Survey in 1977 (Word and Mearns, 1979). Thirteen of the 60-Meter Survey Sites were resampled, and additional reference sites were established at 30 and 150 m in each area, for a total of 38 sites. Grab samples were collected for sediment grain-size, organic material, trace metals, chlorinated and petroleum hydrocarbons, and infaunal analysis. Trawl samples were collected for characterization of megafaunal invertebrate and fish assemblages, and for analyses of tissue contamination and histopathology.

In the 1985 Reference Site Survey sediment types changed with depth from sandy-silt to silty-sand, and organic content increased with shelf depth. Silver (Ag) and Cd were measured in tens and hundreds of ppb (dry wt.) respectively and the other metals (Cr, Cu, Ni, Pb, and Zn) were measured in the tens of ppm range. Trace organic

contaminants (total PAHs, total DDTs, and total PCBs) were measured in the tens of ppb (dry wt.) in sediments. In general, trace contaminants increased in concen- tration with shelf depth and in areas closer to Los Angeles.

Species composition and structure of infaunal assemblages of the mainland shelf were influenced mostly by depth and sediment type (grain-size and organic content). Mainland shelf assemblages at 30 and 60 m sites were dominated by the ophiuroid Amphiodia urtica and polychaete Spiophanes missionensis except in sandy areas where a much different fauna existed. Infaunal assemblages at the 150 m sites were dominated by the polychaete Spiophanes berkeleyorum and A. urtica. Differences in species composition and structure from the shallower sites reflected a transition to normal slope assemblages.

Trawl-caught megafaunal invertebrates were heterogeneously distributed on the mainland shelf. The asteroid Astropecten verrilli, the urchin Lytechinus pictus, and the prawn Sicyonia were the most common and abundant species collected. Similarly, trawl-caught epibenthic and demersal fish were heterogeneously distributed on the shelf. Speckled and Pacific sanddabs, bigmouth sole, and plainfin midshipman were the most common and abundant species collected. At 150 m, both megafaunal invertebrate and fish association showed differences in composition and structure compared with the shallower sites reflecting transitions to the slope fauna. The large amount of variation in the trawl data precluded showing significant trends.

Animal tissues in reference areas were contaminated with lower concentration of chlorinated hydrocarbons than animals collected near sewage outfalls. (For sewage outfall comparison, see Brown, et al., 1986). Livers of four species of flatfish averaged around 4.3 ppm (wet wt.) and hepatopanreas from Sicyonia averaged around 0.61 ppm (wet wt.) in chlorinated hydrocarbon contaminants.

Environmental conditions in the Reference Site Survey are similar to those observed in the 60-Meter Survey. Compared with other studies on the mainland shelf of the region, the Reference Site Survey showed similar trends in sediments an biological assemblages with depth. The reference sites sampled represent "normal" mainland shelf conditions, and may be the least contaminated coastal sites in the region.

Copies of the full report of the Reference Site Survey are available from SCCWRP as:

Thompson B.E., Laughlin J.D., and Tsukada D.T. (1987). Reference Site Survey, 1985. SCCWRP Technical Report #221. 50 pp.

REFERENCES:

Word J.Q and Mearns A.J. (1979). 60-Meter Control Survey Off Southern California. SCCWRP Tech. Memo #229. 58 pps.

Brown D.A., Gosset R.W., Hershelman G.P., et al (1986). Municipal Wastewater contamination in the southern California Bight, I. Metal and Organic Contaminants in Sediments and Animals. Mar. Env. Res. 18, 291-310.

SEWAGE SLUDGE DISPOSAL: NEW APPROACHES TO MONITORING FOR BIOLOGICAL EFFECTS

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In a study of the Thames estuary sewage sludge disposal ground, a number of techniques have been used to investigate biological effects, in conjunction with physico-chemical studies of sludge dispersion. Traditional methods employed included the analysis of macrobenthic abundance and of contaminant body burdens in important species. Other aspects of the biology of these species have also been investigated. These included population biology, disease, and bacteriology, the last forming a link with dispersion studies. Experimental methods, aimed at providing a rapid measure of biological impact, used test organisms in field and laboratory situations. Sublethal stress in implanted mussels was measured by physiological and biochemical tests, while bioassays were carried out on a number of organisms. Although the study is still in progress and therefore incomplete, comparisons show that different techniques may lead to different conclusions. Whilst there was evidence in fish for some contamination, possibly associated with sludge, the principal effect identified by macrobenthic analysis was an increase in faunal density at some sites, probably due to organic enrichment. The more sensitive tests showed the presence of stress in mussels from areas known to be influenced by sludge. Larval bioassays showed that toxic effects of sludge in the water column were short-lived. New methods

under consideration at present include population genetics of indigenous cockles and settlement of planktonic larvae. Whereas no single approach can hope to fulfill all the requirements of monitoring for biological effects, it is intended that some of the new techniques described be routinely incorporated in future monitoring programmes.

MANUSCRIPT NOT AVAILABLE AT TIME OF PUBLICATION

Concordance Between Laboratory Sediment Bioassays and the Characteristics of Benthic Macroinvertebrate Communities in Commencement Bay, Washington

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ABSTRACT

Laboratory sediment bioassays and impacts to benthic macroinvertebrate communities were evaluated at 43 stations in Commencement Bay, Washington (a chemically contaminated embayment) and at 4 stations in Carr Inlet, Washington (a relatively uncontaminated reference embayment). Three bioassays commonly used on the west coast of the U.S. were evaluated.: the amphipod mortality test using Rhepoxynius abronius, the bivalve larvae abnormality test using the oyster Crassostrea gigas, and the Microtox test using the bacterium Photobacterium phosphoreum. Impacts to benthic macroinvertebrate communities were determined using two assessment techniques: statistically significant reductions in the abundances of major taxa (Polychaeta, Mollusca, Crustacea) and numerical classification analyses based on species abundances.

Comparisons between the results of the bioassays and benthic assessments were made using three measures of agreement: concordance, sensitivity, and efficiency. Concordance was represented by the percentage of stations at which the effect/no effect designations of different measures of impact were in agreement. Sensitivity was defined as the percentage of stations with observed effects based on one measure of impact that were predicted as such by a second measure of impact. Efficiency was represented by the percentage of all stations predicted to have effects by one measure of impact that actually had observed effects based on a second measure of impact.

The concordance between impact designations based on the two benthic assessment techniques was significant for molluscs ($P \leq 0.01$) and crustaceans ($P \leq 0.05$), but not significant ($P > 0.05$) for polychaetes. Although the sensitivity of reductions in major taxa to identify effects based on classification analysis was only moderate in magnitude, the efficiency of these comparisons was relatively high.

Of the three bioassays, the oyster larvae abnormality test exhibited the highest level of concordance and efficiency with respect to both benthic assessment techniques, and an intermediate level of sensitivity. By contrast, the amphipod mortality test showed the lowest values for all three measures of agreement. The Microtox test exhibited intermediate levels of concordance and efficiency, but the highest degree of sensitivity.

Results of these comparisons suggest that reductions in the abundances of major benthic taxa can be used to accurately identify areas at which major impacts would be found using classification analysis based on species abundances. Results of these comparisons also suggest that of the three bioassays considered, the oyster larvae abnormality test provides the best predictions of impacts to benthic macroinvertebrate communities.

INTRODUCTION

Laboratory sediment bioassays are used to estimate the biological significance of chemical contamination in aquatic environments. These tests are particularly valuable compared to field surveys of effects on indigenous biota because they are relatively inexpensive and are conducted under controlled conditions. However, a major limitation of many laboratory sediment bioassays is the uncertainty of how the bioassay results correspond to in situ effects on indigenous biota. Knowledge of this relationship is essential for evaluating the ecological meaning of the results of each kind of bioassay.

In this study, we evaluate the relationships of three marine laboratory sediment bioassays with impacts to benthic macroinvertebrate communities in chemically contaminated areas of Commencement Bay, Washington. These data were collected as part of a U.S. EPA Superfund remedial investigation and were used in conjunction with other measures of biological impacts (i.e., fish pathology, bioaccumulation) to identify and rank contaminated areas in the Bay (Barrick et al. 1985).

Because a detailed evaluation of the three bioassays has been presented elsewhere (Williams et al. 1986), it will not be repeated here. Instead, this paper provides a detailed evaluation of the benthic macroinvertebrate communities, followed by a comparison of those results with the bioassay data described by Williams et al. (1986). To our knowledge, this paper provides the most comprehensive assessment available of how selected marine laboratory sediment bioassays correspond to in situ effects on indigenous biota.

METHODS

FIELD COLLECTION

Sampling was conducted during March 1984 at 43 stations in Commencement Bay, Washington (Figure 1) and at 4 stations in Carr Inlet, Washington, a relatively uncontaminated reference embayment. Within Commencement Bay, stations were located in areas having different kinds and degrees of chemical contamination to represent a range of environmental conditions and likely biological effects. Nine stations were located near major sources of contamination (i.e., a combined sewer overflow, pulp mill, chemical plant, and copper smelter) to evaluate the small-scale gradients of contamination and effects related to those sources (Figure 1). The remaining 34 stations in Commencement Bay were located near minor contaminant sources or in areas removed from any known source.

At each station, sampling was conducted concurrently for sediment chemical analyses, sediment bioassays, and benthic macroinvertebrate communities. Synoptic collection of samples for all three kinds of analyses ensured that all

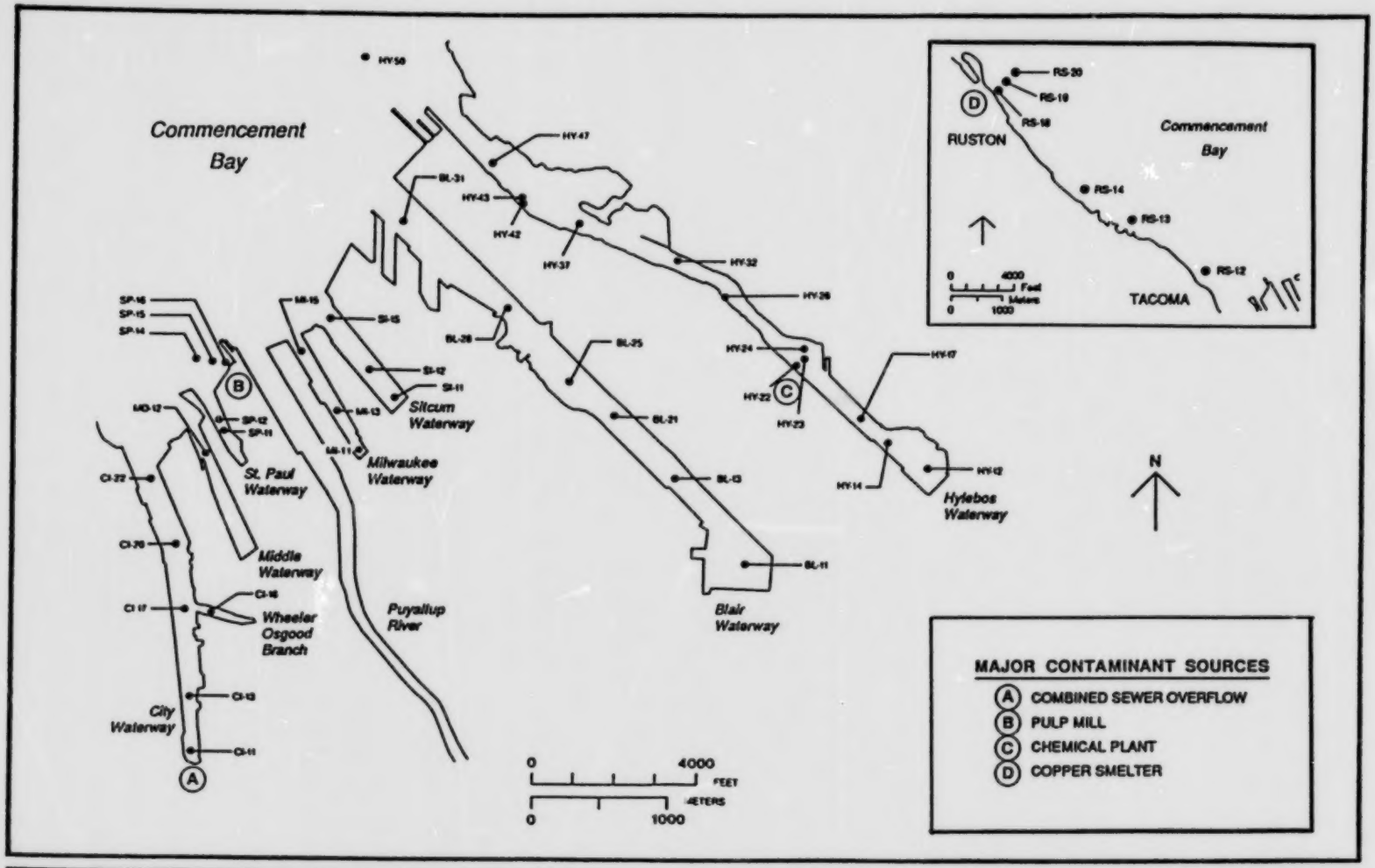

Figure 1. Station locations in Commencement Bay.

results were related as closely as possible on both a temporal and spatial scale.

Sediments for chemical analyses and bioassays were collected using a 0.1-m^2 van Veen bottom grab. The upper 2 cm of sediment was removed from each grab sample and placed in a stainless steel bowl. The sediment was then homogenized by stirring with a stainless steel spoon until the color and texture was visually uniform. Subsamples for bioassays were placed in polyethylene bags and stored on ice. Subsamples for chemical analysis were placed in precleaned jars with teflon cap liners, stored on ice in the field, and frozen upon return to the laboratory. To prevent cross-station contamination, the grab was rinsed with seawater and the subsampling utensils were rinsed in sequence with seawater, methanol, and dichloromethane before each station was sampled.

Sediments for analysis of benthic macroinvertebrate communities were sampled using a 0.06-m^2 van Veen bottom grab. Four replicate grab samples were collected at each station. Each sample was sieved through a stainless steel screen having a mesh size of 1.0 mm. Retained material was transferred to a polyethylene bag, fixed with a 15-percent solution of neutral buffered formalin, and stained with rose bengal.

LABORATORY METHODS

The grain size distribution of sediments was determined using standard sieve and pipette techniques (Folk 1968). Concentrations of U.S. EPA priority pollutant metals and organic compounds were determined according to methods described in Barrick et al. (1985).

Detailed methods of the three laboratory sediment bioassays considered in this study are available in Williams et al. (1986). These tests included the 10-day amphipod mortality bioassay using Rhepoxynius abronius (Swartz et al. 1985), the 48-h bivalve larvae abnormality bioassay using Crassostrea gigas (Chapman and Morgan 1983; ASTM 1985), and the 15-min Microtox bioassay using the bacterium Photobacterium phosphoreum (Bulich et al. 1981; Beckman Instruments 1982). These bioassays have frequently been used to evaluate sediment toxicity along the west coast of the U.S. (e.g., Schiewe et al. 1985; Swartz et al. 1985; Chapman et al. 1985, 1987; Chapman 1986), and represent acute responses by a range of taxa and life stages.

Samples of benthic macroinvertebrates were transferred to 70 percent ethanol within 14 days from collection. Organisms were then sorted from the debris in each sample using a dissecting microscope. Taxonomic identifications were made to the lowest level possible, given the available literature and the physical condition of the organisms.

DATA ANALYSIS

Impacts to benthic macroinvertebrate communities were determined using two methods. In both cases, comparisons were made between communities at potentially impacted stations and those at reference stations. Carr Inlet and Blair Waterway were used as reference areas. Carr Inlet was used as a reference area for stations located along the Ruston Shoreline because both areas are relatively open coastlines with similarly coarse grain-size distributions (e.g., percent fine-grained sediment = 4-23 and 3-33 percent, respectively). Blair Waterway was used as a reference area for the remaining six Commencement Bay

waterways because both kinds of areas are confined manmade waterways with similar grain-size distributions (e.g., percent fine-grained sediment = 37-84 and 26-89 percent, respectively). Although all stations in Blair Waterway were chemically contaminated to some extent, the magnitude of chemical contamination in that waterway was much less than the levels of contamination observed in the other six waterways (Barrick et al. 1985). Blair Waterway was therefore considered a minimally impacted waterway.

The first method used to determine impacts to benthic macroinvertebrate communities involved a normal (Q-mode) numerical classification analysis of the log-transformed abundances of all macroinvertebrate species. The analysis was made using the Bray-Curtis similarity index and a group-average clustering strategy. Two separate analyses were made: one for the Ruston Shoreline and Carr Inlet and one for the seven Commencement Bay waterways.

The second benthic assessment technique involved the determination of statistically significant reductions ($P \leq 0.05$) from reference levels in the log-transformed abundances of three major taxonomic groups: Polychaeta, Mollusca, and Crustacea. Comparisons were made between the abundances at each potentially impacted station and the pooled results from each respective reference area using the t-test. The influence of multiple comparisons on the experimentwise error rate of 0.05 was removed by conservatively adjusting each comparisonwise error rate using Bonferroni's technique (Miller 1981). Reference conditions for each potentially impacted station were stratified by sediment grain size as shown in Table 1.

Three kinds of comparisons were made between the results of the two benthic assessment techniques and between the results of each bioassay and each kind of

TABLE 1. PAIRINGS OF REFERENCE STATIONS AND POTENTIALLY IMPACTED STATIONS USED FOR COMPARISONS OF ABUNDANCES OF MAJOR BENTHIC TAXA

Reference Areas		Potentially Impacted Areas	
Station	Range of Percent Fine-Grained Sediment	Station	Range of Percent Fine-Grained Sediment
BL-28	37	SP-11, SP-12, SP-15 CI-11, CI-22 HY-14	26-49
BL-11 BL-21 BL-31	55-64	SP-14, SP-16 CI-13, CI-16, CI-17, CI-20 HY-12, HY-17, HY-22, HY-28, HY-32, HY-37, HY-42, HY-43, HY-47 SI-11, SI-12 MD-12	55-80
BL-13 BL-25	84-88	HY-23, HY-24, HY-50 SI-15 MI-11, MI-13, MI-15	81-89
CR-11 CR-12 CR-13 CR-14	4-23	RS-12, RS-13, RS-14, RS-18, RS-19, RS-20	3-33

benthic analysis. In the first comparison, the concordance of the effect/no effect designations for all stations was determined. Significance of concordance was evaluated using the binomial test and an expected level of concordance of 0.5 (i.e., that for random agreement).

The two additional kinds of comparisons considered the sensitivity and efficiency of one kind of indicator to predict the results observed for a second kind of indicator. Sensitivity was represented by the percentage of stations with observed effects that were correctly predicted as such. Efficiency was represented by the percentage of all stations predicted to have effects that actually had observed effects.

RESULTS AND DISCUSSION

CHEMICAL CONTAMINATION

To demonstrate the large differences in chemical contamination between Carr Inlet and Commencement Bay, a summary of sediment concentrations of selected contaminants is presented in Table 2. Results for Commencement Bay are presented as elevations above the mean concentrations observed in Carr Inlet.

At least three kinds of chemicals were elevated to $\geq$10 times their respective values observed in Carr Inlet at every study area in Commencement Bay. The largest number of chemicals with elevations $\geq$10 (i.e., eight chemicals) was found at Ruston Shoreline. Half of those chemicals were metals, and were related largely to the copper smelter located at the outer end of the shoreline (Figure 1). The highest elevation of any kind of chemical in Commencement Bay (i.e., 1,300) was found for

TABLE 2. COMPARISONS OF CONCENTRATIONS OF MAJOR CHEMICAL CONTAMINANTS IN COMMENCEMENT BAY WITH RESPECTIVE CONCENTRATIONS IN CARR INLET[a]

Chemical Contaminant	Elevation Above Mean Concentration in Carr Inlet[b,c,d]								Mean Concentration in Carr Inlet[e] (ug/kg dry weight)
	HY	BL	SI	MI	SP	MD	CI	RS	
Organic Compounds:									
LPAH[f]	45*	28*	68**	60**	73**	110***	120***	87**	41
HPAH[g]	120***	42*	65**	68**	27*	97**	140***	85**	79
PCBs[h]	48*	6	8	7	17*	9	12*	19*	6
Chlorinated benzenes[i]	10*	4	3	3	2	6	9	3	U21
Chlorinated butadienes[j]	130***	3	2	1	1	7	2	2	U62
Phthalates[k]	4	3	<1	<1	<1	5	7	5	280
Phenol	6	5	4	2	12*	11*	9	5	33
4-methylphenol	7	12*	10*	13*	1,300***	33*	30*	10*	13
Metals:									
Arsenic	12*	8	11*	4	2	10*	8	620***	3,370
Cadmium	2	2	3	2	2	3	6	27*	950
Copper + Lead + Zinc	10*	5	24*	7	6	18*	22*	120***	35,000
Mercury	8	4	5	4	5	26*	10*	160***	40

[a] Data taken from Barrick et al. (1985).

[b] Elevations are based on the mean of the concentrations observed at all stations in each study area. Mean concentrations were determined using the values of all detected chemicals and the detection limits for undetected chemicals.

[c] Study area abbreviations correspond to those used in Figure 1.

[d] Asterisks denote the following elevations: *≥10, **≥50, ***≥100.

[e] U = Undetected at the detection limit shown.

[f] LPAH = Low molecular weight polycyclic aromatic hydrocarbons (includes naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, and anthracene).

[g] HPAH = High molecular weight polycyclic aromatic hydrocarbons [includes fluoranthene, pyrene, benzo(a)-anthracene, chrysene, benzofluoranthenes, benzo(a)pyrene, indeno(1,2,3-c,d)pyrene, dibenzo(a,h)anthracene, and benzo(g,h,i)perylene].

[h] PCBs = Polychlorinated biphenyls.

[i] Includes 1,2-dichloro-, 1,3-dichloro-, 1,4-dichloro, 1,2,4-trichloro, and hexachlorobenzene.

[j] Includes trichloro-, tetrachloro-, pentachloro-, and hexachlorobutadienes.

[k] Includes dimethyl, diethyl, di-n-butyl, and di-n-octyl phthalates.

4-methylphenol in St. Paul Waterway, and was related primarily to the large pulp mill located at the mouth of the waterway. The lowest level of chemical contamination in Commencement Bay was found in Blair Waterway, where no chemical exhibited an elevation $\geq$50 and only three chemicals (i.e., LPAH, HPAH, and 4-methylphenol) showed an elevation $\geq$10.

Results of the chemical analyses demonstrate that contamination in Commencement Bay was both widespread and patchy. Much of the patchiness was related to the major sources of contamination located in the different study areas. Results of the chemical analyses also support the use of Blair Waterway as representative of minimally contaminated conditions for comparisons of benthic macroinvertebrate communities.

BIOASSAY RESULTS

A summary of the bioassay results from stations in Commencement Bay and Carr Inlet is presented in Table 3. Significant differences ($P \leq 0.05$) from responses in Carr Inlet were found at 34 of the 43 stations (79 percent) sampled in Commencement Bay. Significant responses for all three bioassays were found at eight stations, all of which were located near one of the four major sources of contamination in the Bay. Significant responses for the Microtox test occurred at considerably more stations (29) than did responses for the amphipod mortality and oyster larvae abnormality tests (16 each).

The bioassay results indicate that sediments at most stations sampled in Commencement Bay were toxic to at least one kind of test organism. These results are consistent with the widespread chemical contamination

TABLE 3. SUMMARY OF BIOASSAY RESULTS[a]

Station	Bioassay[b] A[c] (%)	O[d] (%)	T[e] (%)	Station	Bioassay A[c] (%)	O[d] (%)	T[e] (%)
HY-12	16	46*	42*	SP-11	12	21	9*
HY-14	9	25	21*	SP-12	16	30*	14*
HY-17	19	41*	37*	SP-14	100*	100*	95*
HY-22	36*	39*	27*	SP-15	68*	62*	91*
HY-23	26*	36*	17*	SP-16	27*	36*	41*
HY-24	19	25	30*				
HY-28	8	23	5	MD-12	13	24	12
HY-32	20	24	25				
HY-37	18	23	40*	CI-11	52*	63*	38*
HY-42	37*	29	29*	CI-13	20	32*	9*
HY-43	19	21	15*	CI-16	14	32*	8*
HY-47	25	32*	18*	CI-17	17	22	11*
HY-50	16	26	48*	CI-20	30*	32*	3
				CI-22	19	20	-5
BL-11	13	18	5				
BL-13	19	21	14*	RS-12	15	22	-2
BL-21	14	23	8	RS-13	32*	30*	5
BL-25	28*	20	3	RS-14	20	18	4
BL-28	8	16	2	RS-18	95*	70*	27*
BL-31	16	18	37*	RS-19	77*	47*	6*
				RS-20	5	9	62*
SI-11	24	16	20*				
SI-12	31*	18	45*	CR-11[f]	25	17	1
SI-15	25*	16	-4	CR-12[f]	11	11	-1
				CR-13[f]	7	10	1
MI-11	24*	23	2	CR-14[f]	10	14	1
MI-13	14	16	17*				
MI-15	20*	17	30*				

[a] Data taken from Barrick et al. (1985) and Williams et al. (1986).

[b] * $P \leq 0.05$.

[c] A = Amphipod mortality; response = percent mortality.

[d] O = Oyster larvae abnormality; response = percent abnormality.

[e] T = Microtox; response = percent decrease in luminescence.

[f] Reference station.

observed throughout the Bay. The locations of highest sediment toxicity (i.e., where all three bioassays exhibited significant responses) were also consistent with the areas of highest chemical contamination (i.e., near the four major contaminant sources).

Using rank-order comparisons, Williams et al. (1986) found that the responses of the three bioassays showed a high level of overall agreement. However, the authors also noted that substantial differences existed among the responses of the bioassays and may have been attributable to different sensitivities to individual contaminants, different exposure routes, or the heterogeneous distribution of contaminants among stations. The Microtox test appeared to be the most sensitive of the three bioassays, as significant responses for that test were found at almost twice the number of stations at which significant responses were found for the amphipod mortality and oyster larvae abnormality tests.

BENTHIC MACROINVERTEBRATES

The relative abundances of the 10 most numerous taxa in the Commencement Bay waterways, Ruston Shoreline, and Carr Inlet are presented in Figure 2. The benthic macroinvertebrate communities in the Commencement Bay waterways were dominated by two species: the polychaete Tharyx multifilis and the mollusc Axinopsida serricata. Together, these two species accounted for approximately 70 percent of all individuals collected from the waterways. By contrast, no species at Ruston Shoreline or Carr Inlet accounted for more than 15 percent of total individuals. The most abundant species in these two areas were the crustaceans Euphilomedes carcharodonta and Leptochelia dubia, respectively.

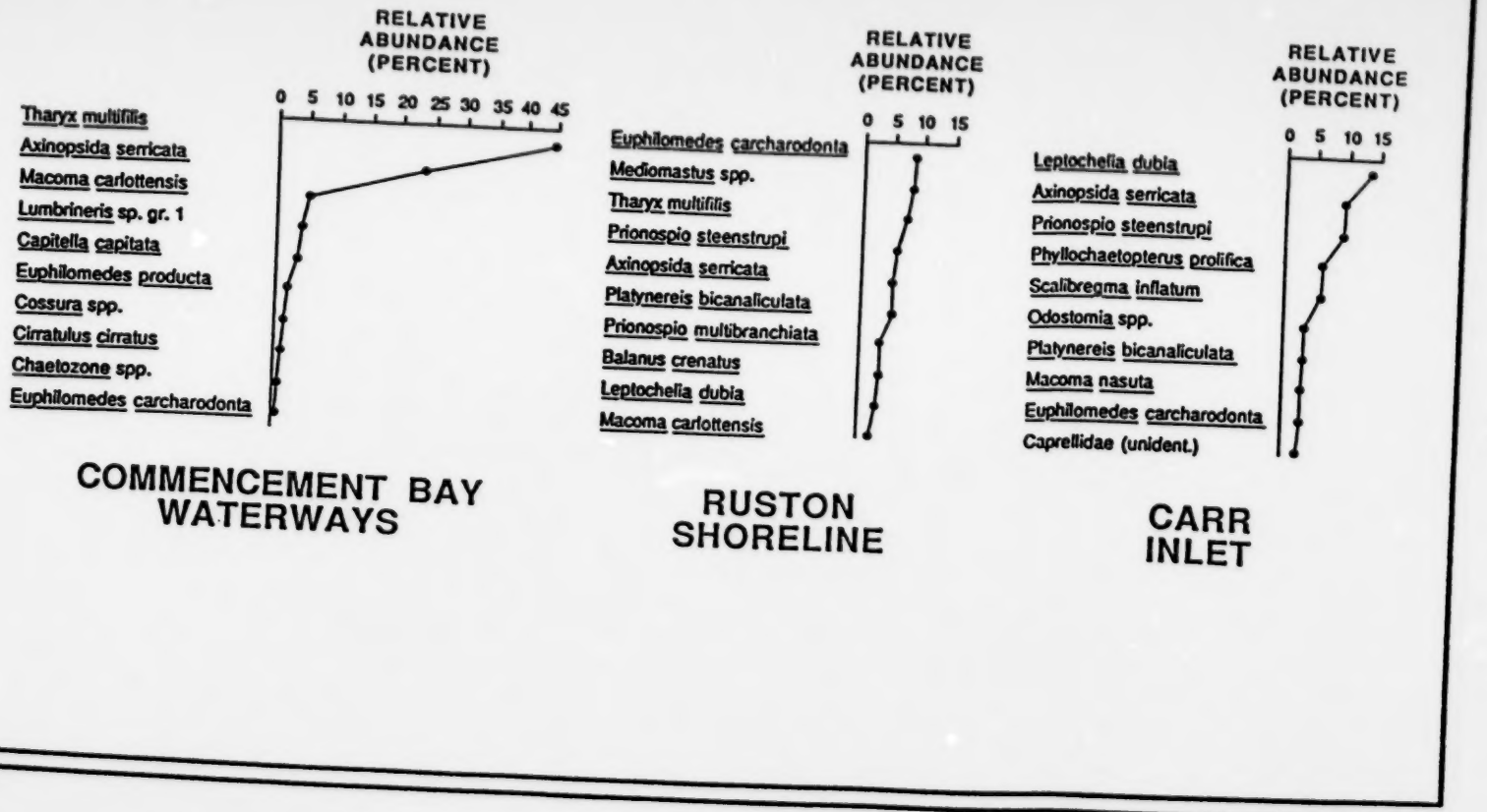

Figure 2. Relative abundances of dominant taxa in each major study area.

The observed pattern of species abundances demonstrates that benthic macroinvertebrate communities in the Commencement Bay waterways were substantially different than those found at Ruston Shoreline and Carr Inlet. The major difference between communities in the three areas is the high dominance of communities in the waterways by only two species. This pattern of dominance is characteristic of highly disturbed communities.

A summary of the abundances of major benthic taxa at stations in Commencement Bay and Carr Inlet is presented in Table 4. Significant reductions ($P \leq 0.05$) in major taxa relative to reference values were found at 17 of the 37 potentially impacted stations (46 percent) in Commencement Bay. Significant reductions in the abundances of more than one major taxon were found at nine stations, seven of which were located near one of the four major sources of contamination in the Bay. The number of stations at which reductions were found for each of the three taxa was lowest for polychaetes (5), highest for molluscs (15), and intermediate in magnitude for crustaceans (10). One or more of the major taxa were absent or nearly absent from three stations: SP-11, CI-11, and RS-18.

Significant reductions of major benthic taxa were distributed throughout Commencement Bay and were most severe (i.e., significant for more than one taxon) in the areas of highest chemical contamination (i.e., near the four major contaminant sources). Compared with the responses of the three bioassays considered in this study, impact determinations based on reductions of major benthic taxa identified only half as many stations as impacted. Polychaete abundances were considerably less sensitive as indicators of chemical contamination than were abundances of molluscs and crustaceans.

TABLE 4. SUMMARY OF ABUNDANCES OF BENTHIC TAXA AND COMPARISONS WITH REFERENCE CONDITIONS

Station	Taxon[a,b] P[c]	M[d]	C[e]	Station	Taxon[a,b] P	M	C
HY-12	1,478	977	140	SP-11	535	187	422
HY-14	1,311	65*	19	SP-12	734	303	103
HY-17	1,554	43*	18*	SP-14	8*	0*	7*
HY-22	843	41*	16*	SP-15	1,002	12*	6*
HY-23	40*	71*	14	SP-16	412	374	75
HY-24	1,038	484	107				
HY-28	3,096	328	204	MD-12	1,404	269	33
HY-32	708	33*	11*				
HY-37	809	127*	53	CI-11	1,470	1*	8
HY-42	2,571	382	111	CI-13	266*	62*	7*
HY-43	1,668	389	86	CI-16	220*	60*	25*
HY-47	276	243*	62	CI-17	2,884	401	139
HY-50	1,910	128	65	CI-20	545	1,378	216
				CI-22	982	988	247
BL-11[f]	642	2,553	272				
BL-13[f]	1,056	1,424	52	RS-12	1,053	474	260
BL-21[f]	943	1,180	132	RS-13	1,049	543	855
BL-25[f]	1,036	1,358	164	RS-14	2,045	154	383
BL-28[f]	251	426	51	RS-18	0*	0*	2*
BL-31[f]	1,478	622	53	RS-19	334	8*	241
				RS-20	419	42*	94
SI-11	1,988	1,676	17*				
SI-12	2,048	682	18*	CR-11[f]	574	332	510
SI-15	2,861	719	66	CR-12[f]	363	263	125
				CR-13[f]	356	210	62
MI-11	1,026	1,145	101	CR-14[f]	285	170	200
MI-13	1,768	1,420	162				
MI-15	1,861	1,368	119				

[a] Numerals represent numbers of individuals per station (0.24 m^2).

[b] Comparisons were made using the t-test. * $P \leq 0.05$ experimentwise.

[c] P = Polychaetes.

[d] M = Molluscs.

[e] C = Crustaceans.

[f] Reference station.

The results of the two classification analyses of benthic macroinvertebrate communities are presented in Figure 3. Also shown in Figure 3 are the stations at which significant bioassay responses and reductions of major benthic taxa occurred.

Using a similarity value of 60 percent to define closely related stations, eight groups were formed. Groups I, V, and VII included all reference stations from Carr Inlet and Blair Waterway. All 17 additional stations included within these groups were therefore considered unimpacted. By contrast, all 20 stations not included in Groups I, V, and VII were considered impacted to some degree.

Groups IV and VIII generally included stations with similarity values less than 60 percent. These two groups should therefore be considered as clusters of outlier stations. All but one of the stations included in these two groups (CI-16) were located near one or more of the four major sources of contamination in Commencement Bay.

Significant bioassay responses were distributed among most of the eight benthic groups. Significant responses for the amphipod mortality and Microtox tests were found at a relatively large number of unimpacted stations (8 and 11, respectively), whereas significant responses for the oyster larvae abnormality test were found at only three unimpacted stations. Significant responses of all three bioassays were largely confined (six of eight stations) to Groups IV and VIII, the two clusters of outlier stations.

Significant reductions in the abundances of major benthic taxa generally were confined to the groups of impacted stations. In only three cases (one for molluscs,

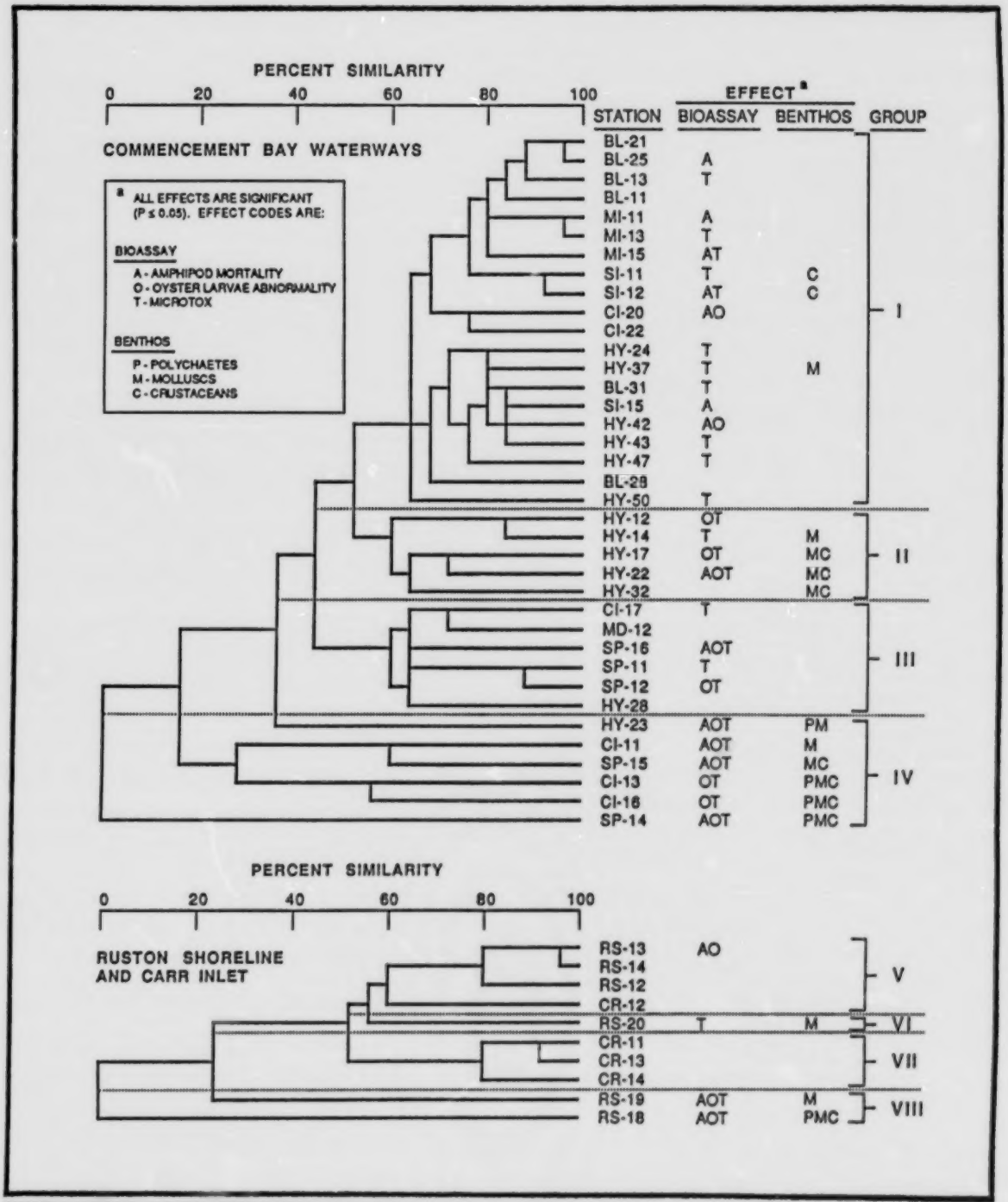

Figure 3. Results of classification analyses based on log-transformed species abundances.

two for crustaceans) were significant reductions in a major taxon found within a group of unimpacted stations. Reductions in all three major taxa were found only in Groups IV and VIII, the two clusters of outlier stations. In addition, all stations with significant reductions in polychaetes were found in these two groups.

Comparisons of the impact designations based on classification analyses, reductions in major benthic taxa, and bioassay responses demonstrated that the three kinds of indicators generally showed similar patterns. The three kinds of indicators exhibited a particularly high level of agreement with respect to identifying the most severely impacted stations.

Comparisons Between Benthic Assessment Techniques

The concordance between impact designations based on reductions in the abundances of major benthic taxa and classification analyses based on species abundances was significant for molluscs ($P \leq 0.01$) and crustaceans ($P \leq 0.05$), but not significant ($P > 0.05$) for polychaetes (Figure 4). The highest level of concordance between the two kinds of assessment techniques (78 percent) was found for molluscs.

The sensitivity of the reductions of major taxa as indicators of effects based on classification analysis was highest for molluscs (65 percent), lowest for polychaetes (25 percent), and intermediate in magnitude (40 percent) for crustaceans (Figure 5). The efficiency of major taxa as indicators was highest for polychaetes (100 percent), lowest for crustaceans (80 percent), and intermediate in magnitude (93 percent) for molluscs (Figure 5).

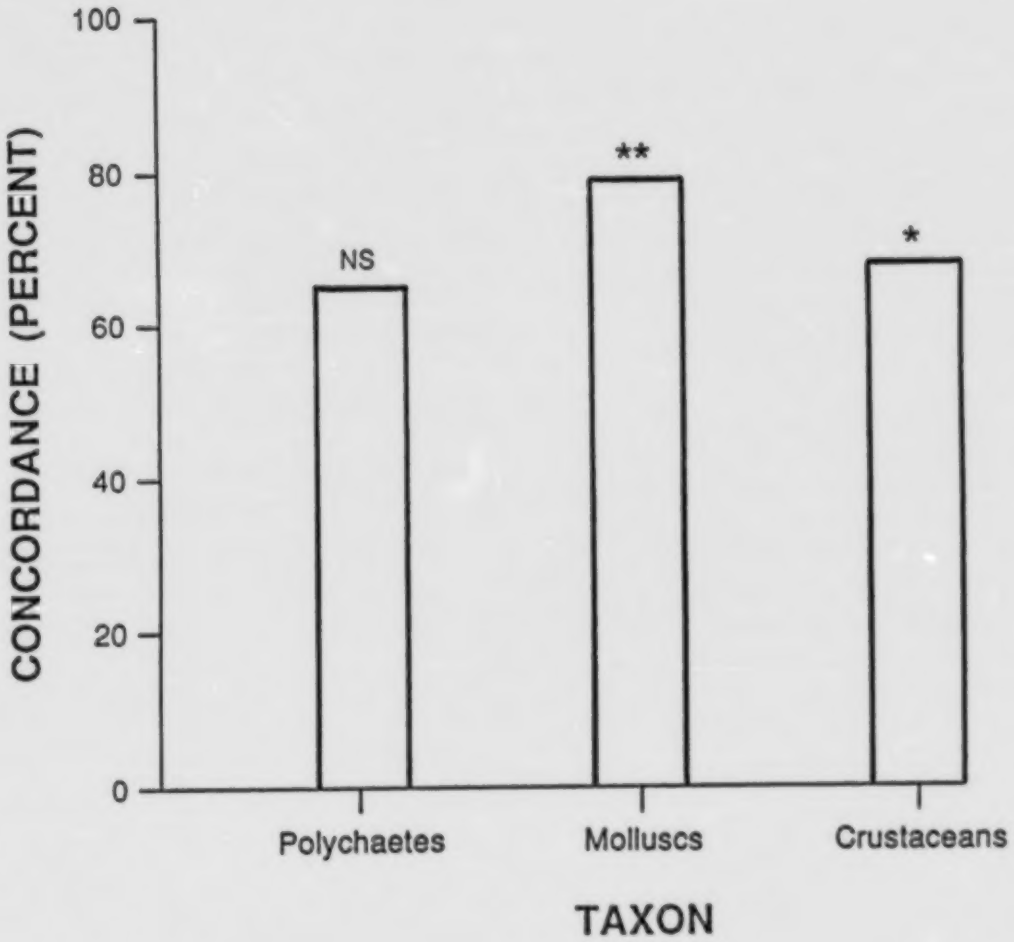

Figure 4. Concordance of impacts based on reductions of major taxa with impacts based on classification analyses.

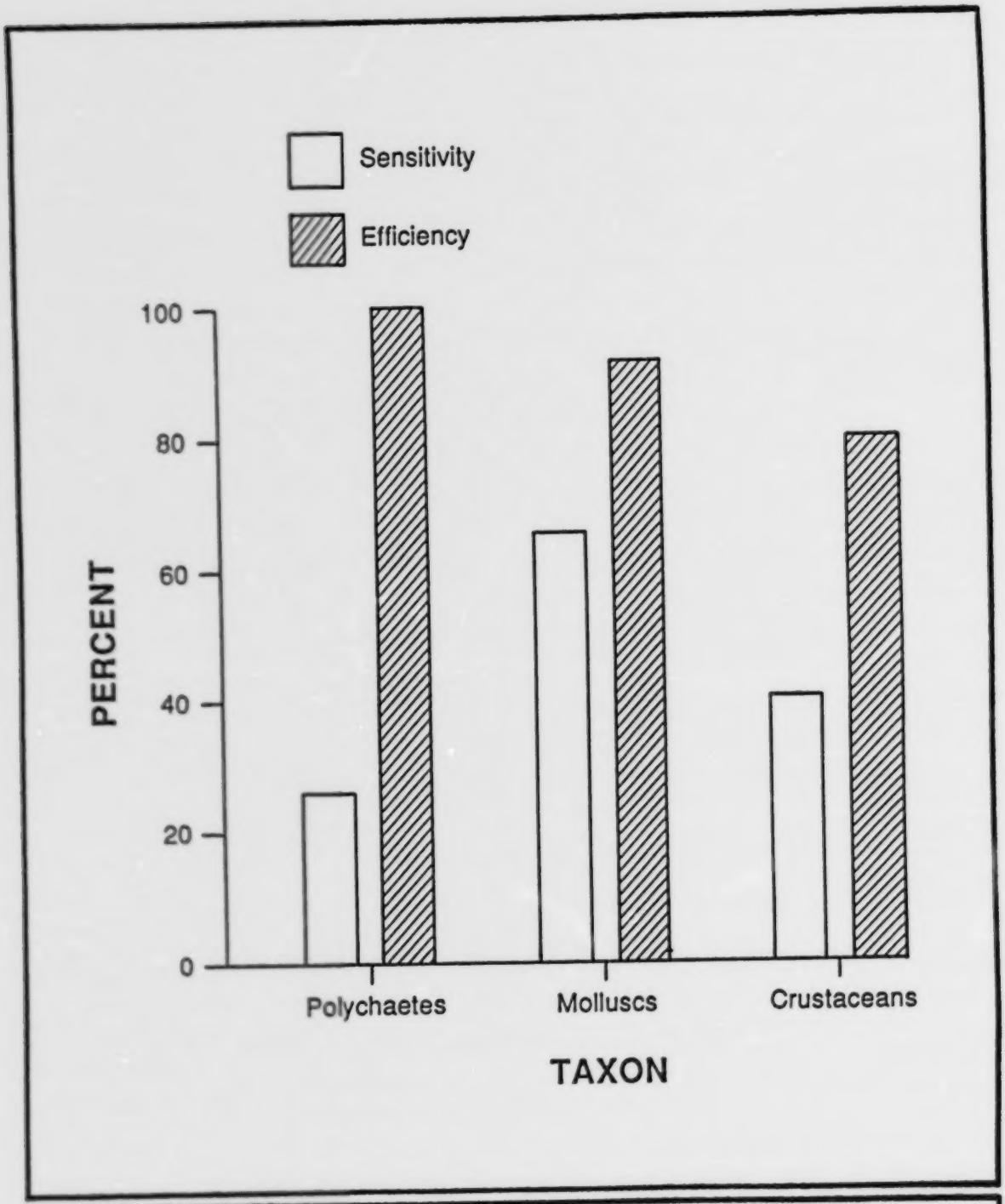

Figure 5. Sensitivity and efficiency of impacts based on reductions in major taxa with respect to impacts based on classification analyses.

Results of these comparisons suggest that agreement between the two kinds of benthic assessment techniques was highest for molluscs. Both concordance and sensitivity were highest for this taxon and efficiency was close to 100 percent. By contrast with molluscs, agreement between assessment techniques was lowest for polychaetes. Although efficiency for this taxon was 100 percent, sensitivity was very low and concordance was not significant.

For all three major taxa, efficiency exceeded sensitivity by a considerable margin (28-75 percent). This pattern indicates that reductions of major taxa were not sufficiently sensitive to identify all stations considered impacted by the classification analyses. However, almost all stations considered impacted by the former indicators were also identified as such by the classification analyses. This pattern is probably attributable in large part to the fact that gross changes in benthic macroinvertebrate communities (i.e., reductions in major taxa) cannot be expected to reflect the more subtle changes in species composition and abundances that influence the results of the classification analyses.

Comparisons Between Bioassays and Benthic Indicators

The concordance between impact designations based on bioassay responses and reductions in the abundances of major benthic taxa (Figure 6) ranged from 41 percent (Microtox/polychaetes) to 78 percent (oyster larvae abnormality/ molluscs). Concordance of results was significant ($P \leq 0.05$) between the oyster larvae bioassay and all three major taxa. By contrast, concordance of results was not significant ($P > 0.05$) between either the

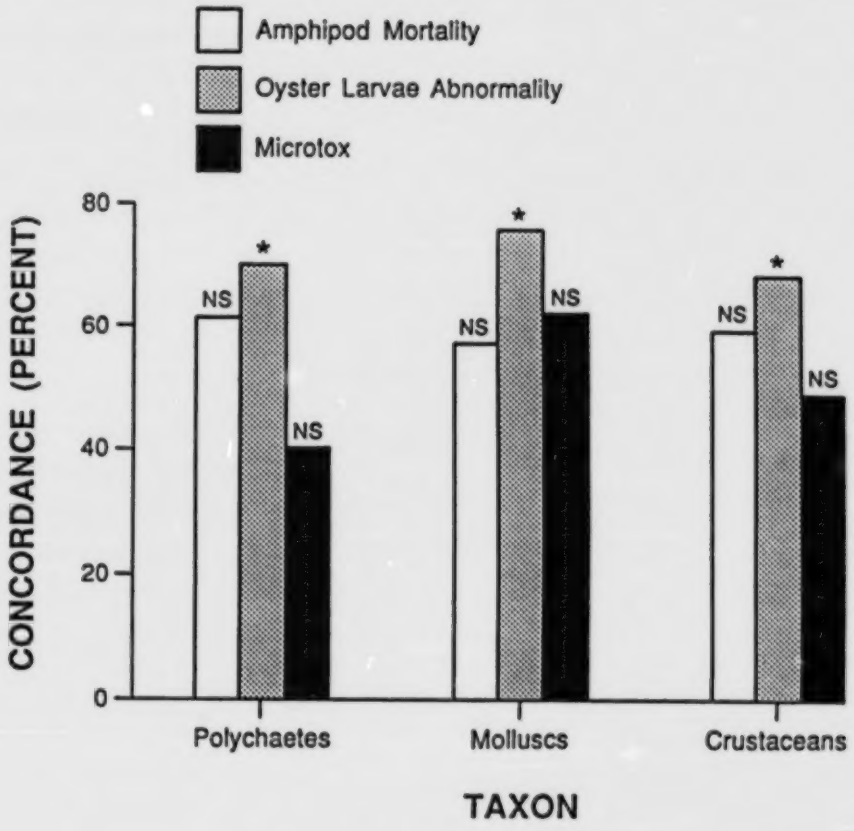

Figure 6. Concordance of bioassay responses with impacts based on reductions in major taxa.

amphipod mortality or Microtox tests and any of the three major taxa.

The concordance of results between bioassay responses and impact designations based on classification analysis (Figure 7) ranged from 57 percent (amphipod mortality test) to 79 percent (oyster larvae abnormality test). Concordance with results of the classification analysis was significant ($P \leq 0.05$) for the oyster larvae and Microtox bioassays, but not significant ($P > 0.05$) for the amphipod mortality test.

The relative patterns of sensitivity and efficiency exhibited by the three bioassays with respect to predicted effects on benthic macroinvertebrate communities were similar for both of the benthic assessment techniques considered in this study (Figure 8). In both cases, sensitivity was lowest for the amphipod mortality test, highest for the Microtox bioassay, and intermediate in magnitude for the oyster larvae abnormality test. Also in both cases, efficiency was lowest for the amphipod mortality bioassay, highest for the oyster larvae abnormality test, and intermediate in magnitude for the Microtox bioassay.

Considering sensitivity and efficiency simultaneously for each bioassay, the amphipod test ranked lowest with respect to both variables. Sensitivity was less than 50 percent with respect to both benthic assessment techniques (40-47 percent), and efficiency was at or only slightly above 50 percent (50-53 percent). The Microtox bioassay exhibited exceptionally high sensitivity (85-94 percent), but moderate efficiency (59-61 percent). By contrast with the Microtox test, the oyster larvae abnormality bioassay showed relatively high efficiency (69-81 percent), but moderate sensitivity (65 percent).

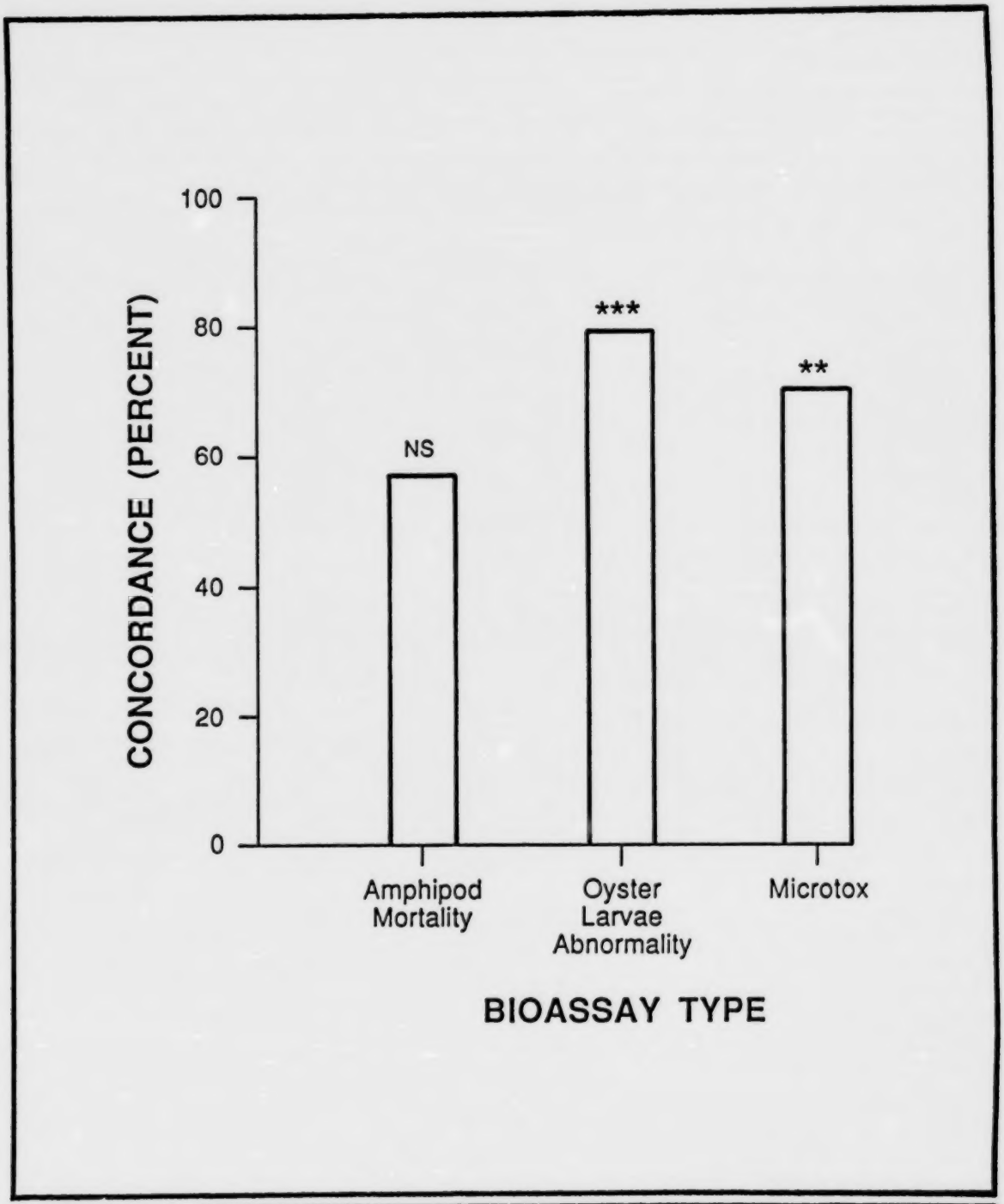

Figure 7. Concordance of bioassay responses with impacts based on classification analyses.

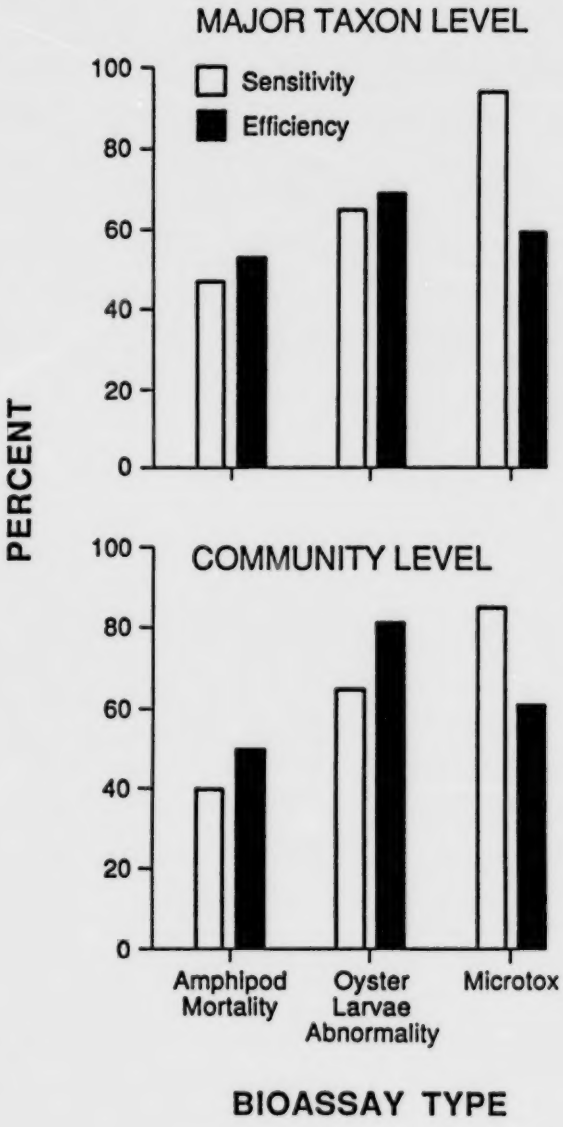

Figure 8. Sensitivity and efficiency of bioassay responses with respect to impacts based on reductions in major taxa and classification analyses.

Results of the comparisons between bioassay responses and the two benthic assessment techniques suggest that the oyster larvae abnormality test exhibited the highest level of agreement with both benthic techniques. Both concordance and efficiency between the results of that bioassay and the patterns identified by both benthic techniques were the highest observed for all three bioassays. Sensitivity was intermediate in magnitude between the values found for the other two bioassays.

The amphipod mortality bioassay exhibited the lowest level of agreement with both benthic assessment techniques. The values of concordance, sensitivity, and efficiency observed for this test were the lowest of those found for the three bioassays.

The Microtox bioassay showed an intermediate level of agreement with both benthic assessment techniques. Values of both concordance and efficiency were intermediate in magnitude with respect to the values found for the other two bioassays. However, the values of sensitivity observed for the Microtox test were the highest of those observed for the three bioassays.

SUMMARY

Chemical contamination was widespread throughout Commencement Bay and most severe near four major sources of contaminants. Significant bioassay responses were found at most stations in the Bay. By contrast, both benthic assessment techniques identified approximately half the stations as potentially impacted. All measures of biological effects generally agreed on the identities of the most severely impacted stations. All of these

stations were located near the four major sources of contaminants in the Bay.

The concordance between impact designations based on the two benthic assessment techniques was significant for molluscs ($P \leq 0.01$) and crustaceans ($P \leq 0.05$), but not significant ($P > 0.05$) for polychaetes. Although the sensitivity of reductions in major taxa to identify effects based on classification analysis was only moderate in magnitude, the efficiency of these comparisons was relatively high.

Of the three bioassays, the oyster larvae abnormality test exhibited the highest level of concordance and efficiency with respect to both benthic assessment techniques, and an intermediate level of sensitivity. By contrast, the amphipod mortality test showed the lowest values for all three measures of agreement. The Microtox test exhibited intermediate levels of concordance and efficiency, but the highest degree of sensitivity.

Results of these comparisons suggest that reductions in the abundances of major benthic taxa can be used to accurately identify areas at which major impacts would be found using classification analysis based on species abundances. Results of these comparisons also suggest that of the three bioassays considered, the oyster larvae abnormality test provides the best predictions of impacts to benthic macroinvertebrate communities.

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THE TOXIC EFFECTS OF POLLUTANTS AND POLLUTED DREDGED MATERIALS ON MARINE ORGANISMS

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INTRODUCTION

Chinese government had published "administration Act for marine Disposal" and become a member of LDC (London Dumping Convention) in 1985. Late the government had announced eight dumping sites in 1987. It was reported that eight big harbours the total amount of dredged materials disposed into the sea was 10^7 m_3 in 1975, and that the industrial sewage and oil-contained waste water disposed along the Chinese coastal area were mounted to several billions and six millions tons, respectively. In the coming years more and more dredged materials and industrial wastes will be dumped in the dumping sites with the increase of population and the development of industry along the coastal area. The toxic effects of Hg, Cd, Cu, Zn, crude oil, diesel oil, dispersant and dredged materials on bivalve (Sinonovacula constricta. Ostrea plicatula Gemlin), Custacea (Artemia SP., Penaeus. penicillatus) and unicellular alga (Chaetoceros calcitrans) have been studied in laboratory in order to provide a scientific bases for the disposal of polluted dredged materials and industrial wastes in the sea. The bioaccumulation of polluted dredged materials was also conducted in the present study.

MATERIALS AND METHODS

Materials

Bivalve (**Sinonovacula constricta, Ostrea uulicillatus** Gemlin,crustacea (**Artemia** SP. **Penaeus. pencillatus**) and unicellular alga (**Chaetoceros calcitrans)** were used in this experiment.

1. The propagation of the straight-hinge larva and juvenile shell of **Sinonovacula constricta**: The fertilized egguus were propagationated by parturifacient method in laboratory. After about 40 days cultivation, the larva developed into juvenile shell (0.7-1.0 mm).

2. The propagation of the straight-hinge larva of **Ostrea plicatula** Gmelin: The larva were propagated either by indoor parturifacient method or by artificial fertilization of the direct analogy of parent clams. The parent clams used were sampled from the sea.

3. The metonauplius of **Artemia** SP.: The winter eggs of **Artemia** SP. obtained from commercial use were incubated in a temperature about 25oC. After 20 hours, they developed into metanaunlius. Then these metanauplius were continually cultured for 12-24 hours.

4.Mysis stage: Mysis stage were collected from the mysis hatching field in our Institute during the reproduction season.

5. Unicellular Alga: Unicellular alga (Chaetoceros calcitrans) were cultivated in laboratory before the toxic test. They were incubated with N-20 ppm, P-2 ppm, Si-0.5 ppm, and Fe-0.1 ppm.

6. Oil: Crude oil was obtained from Shanghai oil field. Diesel No.O was purchased commercially. Crude oil dispersant was from BP Company in UK.

7. $HgCl_2$ $CuSO_45H_2O$, $ZnSO_47H_2O$, $CdCl_2Pb(NO_3)_2$: They were all AP grade with except of $Pb(NO_{3)}$ with CP grade.

8. Dredged materials: They were collected from contaminated sediment near the draining entrance of Xiamen Harbour. The control sediments were collected from Xiamen Navigation Way far away from polluted sources.

Methods

Experimental Conditions:

1. Water for experiment: Both of sea water and fresh water were filtered through sands and then through 0.45 um micropore uuilter. Thus the water was prepared in different degrees of salinity according to the used organism habitats.

2. Preparation of solutions with high toxic concentration: solutions with high toxic concentration prepared freshly or stored in 4oC refrigerator were prepared according to the toxicity of compounds and then diluted into different concentrations before the toxic test. The preparations were as follows:

(1) Oil:

A. The soluble fractions of crude oil and diesel oil: Oil and prepared water were mixed (1:9,V:V) and then shaken (243 times $min.^{-1}$.) for 4 hours. The mixture was put in still state for 24 hours and the water phase was seperated as a test solution.

B. The mixture solution of crude oil and dispersant: It was prepared by stirring the mixture of crude oil and dispersant (2:1,V:V) with 100 mL water with given degrees of salinity by stirring. The solution was put in still state for 12 hours and the water phase was seperated as a test solution.

The test solution was generally prepared before use and the concentration in it was ditermined by a UV Spectrophotometor = 225 um).

(2) Preparation of heavy metal solutions: The heavy metal compounds were dissolved in distill water. The concentration of Hg, Cu, Zn, Cd and Pb in each solution was 1, 1, 1, 10 and 10 mg L^{-1} respectively.

(3) Dredged materials: Both of the polluted and control dredged materials were sifted through an 1 mm-nylon sifter. One part of dredged material was mixed with four parts of sea water on a value bases. The mixture were stirred vigorously for 30 min. and then seperated into three phases after 1 hour of undisturbed settling: the solid phase, the particulate phase and the liquid phase which was obtained by

the filtration of the particulate phase through an 0.45 um filter. Separate bioassays were performed on each of the three phases.

3. Containers and density of organisms: All of the experiments were performed in 400 mL beakers except for the tests of unicellular ALuua which were performed in 500 mL beakers. The population of organisms in beakers were: unicellular alga, 350,000 mL^{-1} in a total volume of 300 mL; straight-hinge larva of bivalve, 10 mL^{-1}; juvenile shell, 0.075 $mL,^{-1}$; metanauplius of **Artemia** SP. 0.125^{-1} and mysis stage, 0.025 mL^{-1}.

All of the experiments were carried out in a constant temperature. Salinity, pH, light flux and concentration of toxic compounds for each experiment were showed on Table 2 in detail.

EXPERIMENTS

1. Oil: The toxic test of dispersant, oil + dispersany and the water soluble fractions of oil and diesel on bivalve (larva of **Sinonovacula constricta** and **Ostrea plicatula** Gmelin), crustacea (metanauplius of **Artemia** SP.) and unicellular alga (**Chaetoceros calcitrans**) were carried out with five different concentrations individually, and the tolerance test of organisms for toxic heavy metals were also observed (Table 3)

2. Heavy metals: Apart from the orthogonal test, the toxic test of Hg, Cu, Zn, Cd, and Pb on bivalve (larva of **Ostrea plicatula** Gemlin), crustacea (metanauplius of **Artemia** SP.) and unicellular alga (**Cheatoceros calcitrans**) were carried

out with five different concentrations individually, and the tolerance test of organisms for the toxic metals were also observed (Table 3)

3. Dredged materials: Bioassay of dredged materials, using the juvenile shell of **Sinonovacula constricta,** was performed in four different concentrations of each phase solution among the three phases. The effects of three phases on organisms were observed too. The experiments were conducted as follows:

(1) Liquid phase: 0% (control), 50%, 100%;
(2) Suspended particulate phase: 0% (control), 50%, 100%;
(3) Solid phase: 0% (control), 30%, 100%.

Bioassays of liquid and suspended particulated phase were performed for 48 hours, and solid phase for 10 days. For the solid phase bioassay, after 10 days the contents in each beaker were washed through a nylon screen with sea water. The organisms were retrieved from the screen and the number of survivers were counted.Organisms surviving in the solid phase bioassay were transferred to a clean, sediment-free water for 2 days to avoid fed sediment in their guts, and then washed with freshwater, dried at 60oC digested in a Teflon vessel. The concentrations of heavy metals accumulated were determined with a Flameless Atomic Adsorption Spectrophometry (703-type, P.E. Co., U.S.A.). All of the above experiments were repeated more than two times. Replicate samples were detected in each measurement. During the test, the feeding, activity and death of oruuanisms were observed under a microscope every 4 hours, and the liquid phase containing pollutant were replaced every 48 hours with the same concentration solution

except for the group of dredged material test in which the liquid phase were replaced with fresh sea water at the same time. Meanwhile, food was also added to each beaker The amount of food added was referred to Philippine's Feeding Experiment of Straight-hinge Larva of Clam.

Based on the mortality or other unnormal habits of organisms tested, sublethal concentration (LC50) was calculated statistically and the sensitivity of organism to the pollutant was estimated. The calculation was performed by PBP-II computer with the statistic method described by Zuo Haijing **et. al.**(1983).

RESULTS

In the toxic test of water soluble fractions of dispersant, oil + dispersant, oil and diesel on juvenile of bivalve and crustacea, the sublethal concentration was statistically calculated based on the mortality of tested animals, as shown in Table 3. The sensitivity of bivalve and crustacea to these toxicity was: dispersant > water soluble fractions of diesel > water soluble fractions of crude oil > crude oil + dispersant. It was observed under microscope that there was a brownish-grey loaf along the pancreas. This change in animal was resulted from the bioaccumulation. During the experiment of water soluble fractions of crude oil and diesel the size and colour of this brownish-grey loaf in these groups was greater than those of control and dispersant added groups. The size and colour of the loaf were related to the experimental time and the toxic concentration. The loaf appeared in the second day in both of high and low concentration groups but deeper colour in the high concentration groups and lighter

in the low concentration groups in which the colour increasement developed with time and stopped as the colour reached to certain degree.

The toxic effects of five heavy metals on crustacea, bivalve and unicellular alga were shown in Table 3, indicating that the heavy metals greatly affected on the tested organisms and the mortality increased with the elevation of the concentration of heavy metals. It was shown from the comparation of 48h-96h LC50 that the sensitivity of organisms tested to heavy metals was Hg,> Cu > Zn > Pb > Cd. The highest tolerance LC50 of organisms tested to Hg and Cu was unicellular alga: 0.049 mg L^{-1} for 48h and 0.086 mg L^{-1} for 96h to Hg and 0.244 mg L^{-1} for 48h and 0.316 mg L^{-1} for 96h to Cu. The lowest tolerance LC50 to Hg was mysis stage (0.006 mg L^{-1} for 24h, 0.002 mg L^{-1} for 48h), and to copper, the straight-hinge larve of **Ostrea plicatula** Gemlin (0.1 mg $^{-1}$ for 48h, 0.005 mg L^{-1} for 96h). The tolerance LC50 to Zn, Pb and Cd was basically the same as those of Hg and Cu.

It was observed under reversing microscope that in high concentration groups a secrete juice was discharged by the straight-hinge larva of **Ostrea plicatula** Gmelin if it was in a toxical condition. The more the secrete juice, the more the death of larve. However, this was not observed during the test of mysis stage. The toxic of Hg, Cu, Zn, Cd and the mixture of these four metals to unicellular alga showed that the toxicity of a single heavy metal was greater than that of the mixture (Fig. 1-6).

Liquid, suspended particulate and solid phases bioassays of dredged material showed that there was not any effects on the surviving of juvenile shell of **Sinonovacula constricta**, but the content of heavy metals in the shell larva in the solid phase group was higher than that of the control (Table 4).

DISCUSSION

It is shown from the results that the toxicity of dispersant,water soluble fractions of crude oil and diesel, crude oil with dispersant added to organisms varied with their concentrations. Lonning **et. al.** (1976) indicated that the toxic effect of dispersant 9527 on the fertilization rate and embyro rate of urchin varied with its concentration: the fertilization rate could not reach to 100% when the concentration was 10 ppm 50% when 10-20 ppm. However, the fertilization was throughout inhibited then the concentration was 30-50 ppm and its effect on the developement of enbryo was also sensitive: the embryo could develope into a juvenile when the dispersant concentration was 1-10 ppm but the larva was generally distortional. Studies on the effect of water soluble fractions of crude oil on urchin (**Melitta quinquiesnerforata**) eggs, Nicol **et. al.** (1977) showed that there was a slight effects on the surviving of urchin juvenile if the oil content was 4%, 50% of the juvenile developed incompletely or did not develope at all and the surviving juvenile developed slowly comparing with that of the control group, if the oil content was 10%. Furthermore, almost no development or all distortional development if any were observed when the content increased to 20%. Our results show many similarities to those described by Lunning **et. al.** (1976) and Nicol **et. al.** (1977).

Byrne et. al (1971) indicated that the toxicity of water soluble fractions of refined oil was greater than that of water soluible fractions of crude oil, using the embryo and juvenile of **Mereenaria** SP. as a monitoring species. After studing the effect of crude oil added with dispersant and water soluble fractions of crude oil on the nitrate

generation of microfauna and microflora fractions of crude oil greater than crude oil added with dispersant (16 X 6.25%) and dispersant alone. However, the toxicity of water soluble fractions Oil diesel was greater than that of crude oil. Our result indicated the toxicity of tested compounds was as follows: Dispersant > water soluble fractions of fuel > water soluble fractions of crude oil > crude oil added with dispersant. These results were basically accordance with those described by Byrne (1971) and Beryl Harty (1982). The only discrepancy was that toxicity of dispersant was greater than that of three other compounds in our results but smaller than that of crude oil and water soluble fractions of crude oil in the result of Beryl Harty. This discrepency may be resulted from the different compositions in the dispersants used in these two experiments.

It has been reported that either single heavy metal or mixture of heavy metals did have some toxic effects on organisms and the effect varied with concentrations. Benoid (1976), Wilber (1971), Nagabhushanam (1986), Riisgaard (1986), Chen Yanuueng (1985) **et. al.** had studied the toxic effects of Hg, Cu, Pb, and Cd on fish, crab, shrimp and alga and showed that the organisms poorly developed if the concentration of heavy metal was low while the organisms died if the concentration was high. In our experiment it was found that true toxic effect Hg, Cu, Zn, Cd and Pb on the juneniles **Penaeue Penicillatus, Artemia** SP., **Ostrea nlicatula** Gmelin and **Chaetoceros calcitrans** were sensitive, and the sublethal concentrations were 0.002-132.4 mg L^{-1} for 48 hours and 0.002-66.22 mg L^{-1} for 96 hours (Table 3).

The toxic effects of Hg, Cu, Zn or Cd and their mixture on **Chaetoceros calcitrans** in present study showed that Chaetoceros caicitrans would die within 24 hours if the concentration of Hg or Cu was 1 mg L^{-1}, whereas the mixture of the four heavy metals in the same concentration (1 mg L^{-1}) could not cause all of the tested alga to death within 6 days (Fig. 1-6). The reason for this may lie in the interaction effects of the four elements. It was proposed (Cheng Zhengfeng, 1985) that each of the four heavy metals or their mixture could result in a synergism as well as a interaction effect in study about the toxic effect of heavy metals on the growth and the chlorophyll content of **Cheatoceros muelleri** Lemmermann.

Many works have been done on the toxic effect of dredged material on bionics. After studying the biological fate of dredged material dumped at sea, Marquenie **et. al.**, indicated that heavy metals in dredged material effected filtration animals. Kester **et. al.** (1983) reported the biological effect of dredged material, and exhibited that the largest effect was found in the solid bioaccumulation. Our result similarily showed that the survival rate of juvenile shell of **Sinonovacula constricta** exposed to the liquid, suspended particulate and solid phase of dredged materials were more than 90%, and that the bioaccumulation occurred in shell larve from solid phase (Table 4).

It is shown on Table 4 that Cd concentration in the polluted dredged material is low, but the value of Cd accumulated in the body of juvenile shell exposed to the dredged material is high (77.6 mg L^{-1}). It is proposed that Cd could accumulated in shell in the same way of the accumulation of Ca, as the crust was not removed by the analysis. The bioaccumulation of Cu and Zn is also high.

This is caused by the filtration of the filtration animals in the water with high concentration of heavy metals dissolved from the polluted dredged material.

It is apparent that the concentration of each heavy metal accumulated in shell larve changed with the content of these elements in the control sediment and the dredged material (Table 4).

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Table 1 . Orthogonal layout

No. \ Factor / Level	A Hg(PPM)	B Cu(PPM)	C Zn(PPM)	D Cd(PPM)
1	1(1PPM)	1(1)	1(10)	1(20)
2	1	2(0.1)	2(0.1)	2(10)
3	1	3(0.01)	3(0.01)	3(1)
4	2(0.1PPM)	1	2	3
5	2	2	3	1
6	2	3	1	2
7	3(0.01PPM)	1	3	2
8	3	2	1	3
9	3	3	2	1

Continue table 3. Lc_{50} of organisms to toxics

Toxic	Cd				Pb	
Concn otoxic (mgL^{-1})	0.001, 0.01, 0.1, 1, 10.				0.001, 0.01, 0.1, 1, 10, 100.	
Organism	Artemia sp		C. Calcitrans		P. pencicllatus	
Time	48	96	48	96	24	48
$Lc_{50}(mgL^{-1})$	44.16	99.54	132.434	40.55	15.849	1.995
Reliability of Lc_{50} (mgL^{-1})	30.1-72.6	6.644-14.93	91.961-190.72	32.108-51.214	7.954-31.579	0.599-6.648

Table 2. Testing conditions of organisms

Species		Temp. °C	PH	Gravity of sea water	Light flux LX
Sinonovacula constricta	Straight-hinge Larva	23	8.1	1.012	1000-2000
	Juvenile shell	23	8.1	1.020	1000-2000
Penaeus penicillatus Mysis stage		24	8.1	1.020	1000-2000
Ostrea plicatula straight-hinge Larva		23	8.1	1.020	1000-2000
Artemia. sp. metanauplius		25	8.1	1.020	1000-2000
Chaetoceros calcitrans		25	8.0	1.015	2000-3500

Table 3. Lc_{50} of organisms to toxics

Toxic	Dispersant				Water soluble fractions of crude oil				Water soluble fractions of diessel			
Concn otoxic (mgL^{-1})	o.o1, 0.1, 1, 10, 100,				0.009, 0.09, 0.9, 4.5,9				0.087,0.87, 8.7, 43.5,8.7,			
Organism	S. constricta		P. penicillatus		S. constricta		O. plicatula		S. constricta		O. plicatula	
Time	48	96	24	48	48	96	48	96	48	96	48	96
$Lc_{50}(mgL^{-1})$	10	0.05	1	0.794	16.005	0.402	2.807	0.658	7.754	0.22	2.11	0.024
Reliability of Lc_{50} (mgL^{-1})	9.56-10.45	0.048-0.053	0421-2.373	0.33-1.91	1512-16.52	0.38-0.43	2.73-2.89	0.631-0.686	7.401-8.12	0.204-0.23	2.035-2.187	0.023-0.024

Toxic	HC								oil dispersomt	
Concn otoxic (mgL^{-1})	0.0001, 0.001, 0.01, 0.1, 1								0.173,1.73,17.3 86.5, 173	
Organism	Artemia sp		P. penicillatus		O. plicatula		C. calcitrans		P. penicillatus	
Time	48	96	24	48	48	96	48	96	48	96
$Lc_{50}(mgL^{-1})$	0.019	0.006	0.006	0.002	0.032	0.01	0.049	0.086	25.12	7.995
Reliability of $Lc_{50}(mgL^{-1})$	0.014-0.025	0.004-0.01	0.003-0.013	0.001-0.004	0.032-0.032	0.01-0.01	0.036-0.066	0.059-0.124	15.71-39.404	2.52-22.66

Continue table 3. Lc_{50} of organisms to toxics

Toxic	Cu							
Concn. otoxic (mgL^{-1})	0.001, 0.01, 0.1, 1, 10,							
Organism	Artemia sp		P. penicillatus		O. plicatula		C. calcitrans	
Time	48	96	24	48	48	96	48	96
$Lc_{50}(mgL^{-1})$	2.188	0.38	0.126	0.016	0.1	0.005	0.244	0.316
Reliability of Lc_{50} (mgL^{-1})	1.508-3.175	0.267-0.541	0.055-0.287	0.005-0.05	0.096-0.104	0.0045-0.0055	0.177-0.337	0.316-0.316
Toxic	Zn							
Concn. otocic (mgL^{-1})	0.001, o.o1, 0.1, 1, 10.							
Organism	Artemia sp		P. penicillatus		O. pliactula		C. calcitrans	
Time	48	96	24	48	48	96	48	96
$Lc_{50}(mgL^{-1})$	4.295	1.096	0.1	0.05	1.995	0.632	66.22	18.113
Reliability of Lc_{50} (mgL^{-1})	3.012-6.126	0.722-1.666	0.076-0.132	0.016-0.154	1.939-2.053	0.6-0.666	46.611-94.082	10.684-30.709

Table 4. The bioaccumulation of dredged materials on the juvenile of S constricta

Dredged material	Heavy metal content in tested organism after 10 days (mg kg^{-1})					number of organisms	
	Hg	Cu	Zn	Pb	Cd	begin	end
Polluted 100%	0.25	17.4	77.3	3.75	77.6	30	28
Polluted 30%	0.24	17.5	44.6	3.73	3.65	30	27
Control 100%	0.16	11.1	36.4	2.3	1.76	30	28
Metal content in the polluted	0.58	327	1852	139	8.9		
Metal content in the control	0.09	21.18	80.16	34.83	0.06		

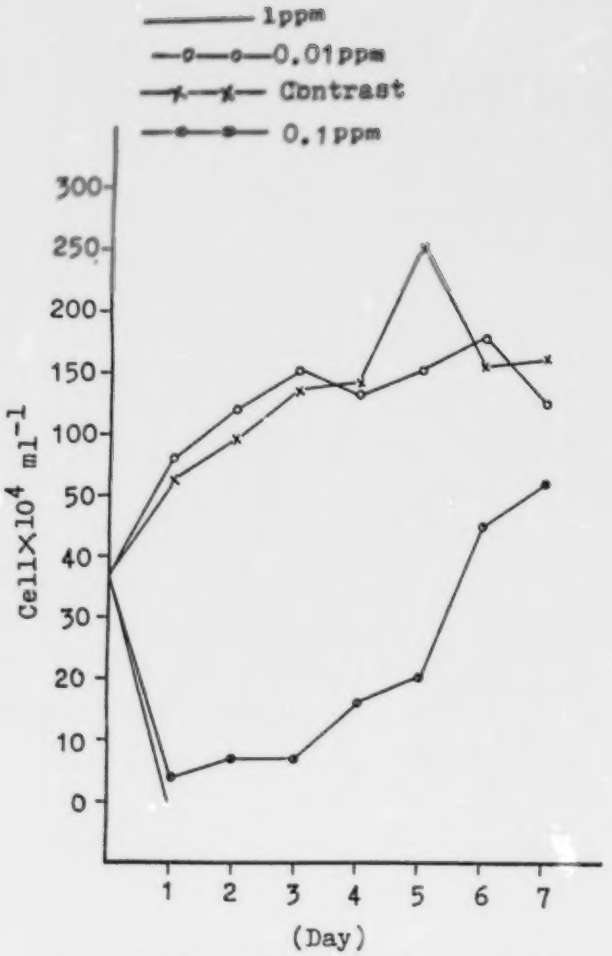

Fig.1 The effect of Hg concentration on the reproduction of Chaetoceros calcitrans.

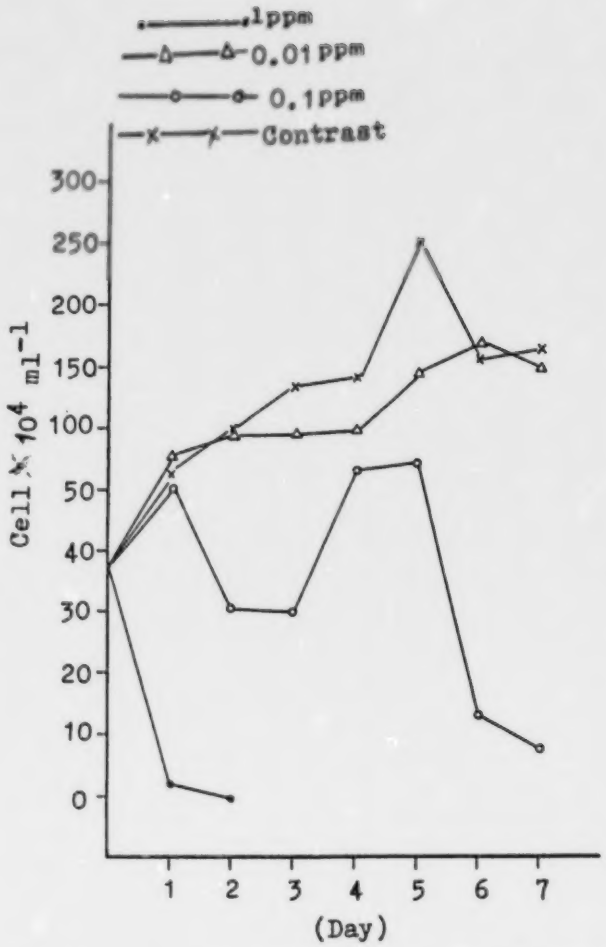

Fig. 2 The effect of Cu concentration on the reproduction of Chaetoceros calcitrans.

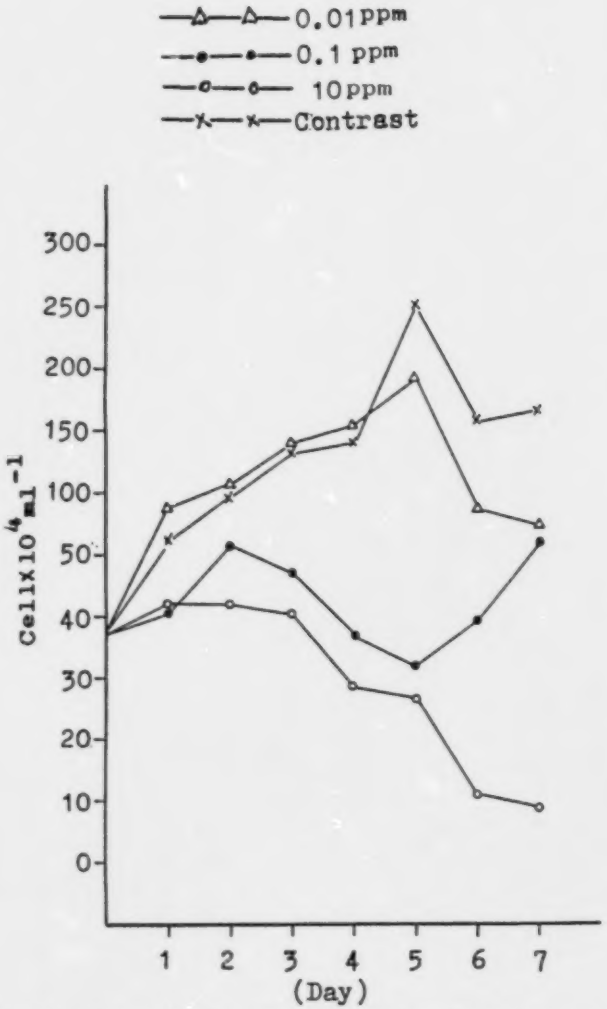

Fig. 3 The effect of Zn concentration on the reproducrion of Chaetoceros calcitrans.

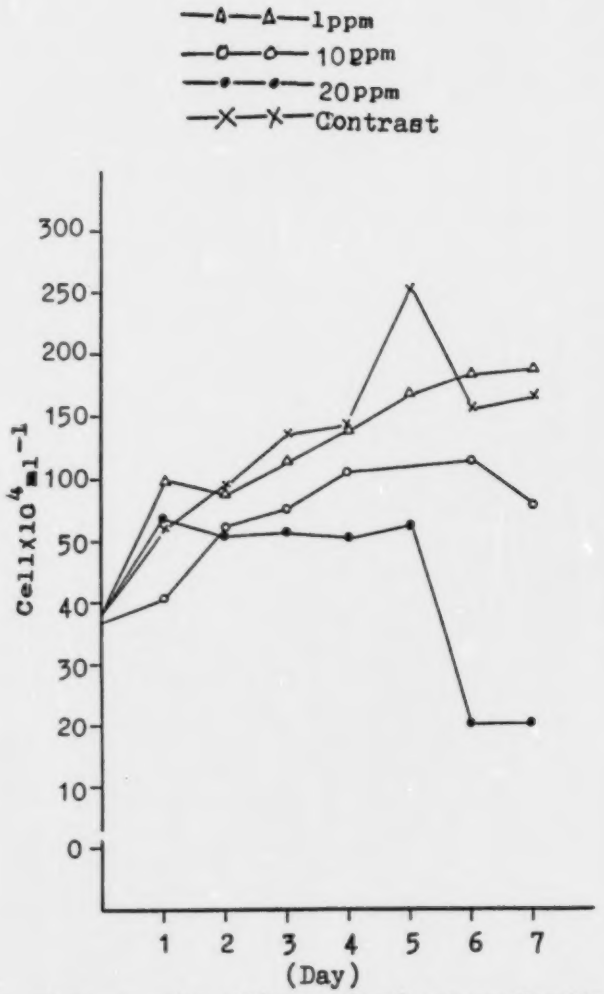

Fig. 4 The effect of Cd concentration on the reproduction of Chaetoceros calcitrans.

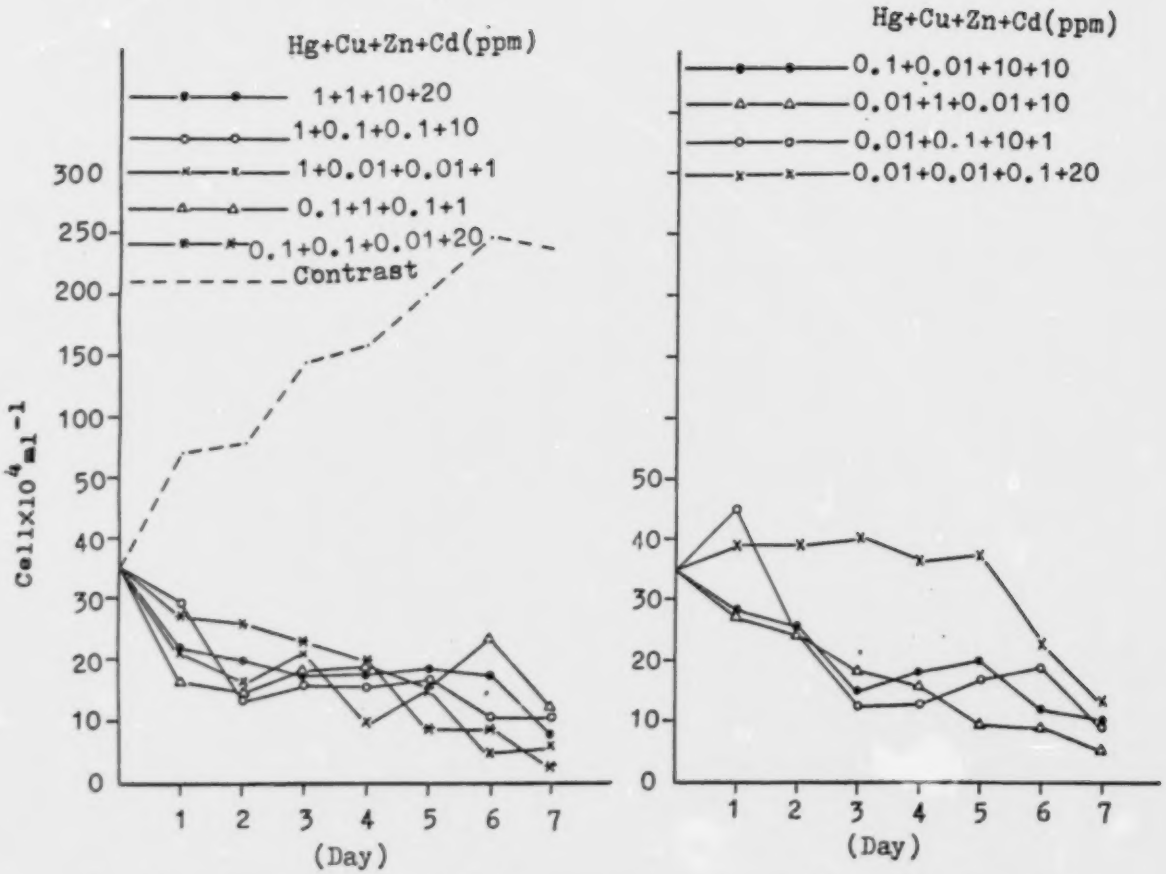

Fig. 5-6 The effects of orthogonally mixed heavy metals Hg,Cu, Zn and Cd on the reproduction of Chaetoceros calcitrans.

THE ROLE OF BACTERIAL BIOASSAYS IN PREDICTING THE POTENTIAL IMPACT OF DUMPING CONTAMINATED SEDIMENTS IN THE OCEAN

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ABSTRACT:

Sediment bioassay procedures using radiolabelled substrates and enzyme specific substrates provide a rapid, sensitive and economical means of assessing the potential toxicity of pollutants to indigenous microflora.

INTRODUCTION

Coastal marine sediments are important sites for nutrient regeneration, the transformation of trace metals and the degradation of organics. Since these processes are primarily mediated by microorganisms (Bauer et al., 1981; Haines et al., 1982; Lee et al., 1985), much research has been conducted on the environmental impact of contaminants on microorganisms and on the use of bacteria in bioassay tests (Bitton, 1983). Bacteria are ideally suited as test organisms in routine bioassays since they grow rapidly, have short life cycles, are inexpensive to maintain, and have enzymatic processes common to higher organisms (Liu, 1981).

Use of Single species vs. community tests in bioassays

To predict effects in nature, it is not sufficient to show a physiological change in a laboratory culture, but the ecological effect of that change on the community and ecosystem must be demonstrated (Bayne, 1981). Although some researchers have argued that the use of pure cultures avoids the "anomalous results" attributable to the different sensitivities to contaminants by organisms in mixed cultures (Thompson and Watling, 1985), the environmental impact of a toxicant can not be predicted with a single species test (Cairns, 1984). While the use of a wide spectrum of species specific tests may increase reliability and simplify interpretation of data (Coleman and Qureshi, 1985; Vives-Rego et al., 1986), future efforts to predict pollutant impact should be based on higher levels of

organization and diversity than a single species in isolation (Ausmus, 1984; Neuhold, 1986).

The standard microorganisms used in single species tests, such as Photobacterium sp. in the Microtox bioluminescence test (Bulich, 1979), Spirillum volutans in the bacterial mobility test (Dutka et al., 1983), and Samonella typhimurium in the Ames test (Ames et al., 1975), are often foreign to the area of concern and therefore, results from tests with these organisms may not represent the response of the indigenous biota to toxicants. In addition, laboratory species are often those that are most easily maintained and therefore, the most resistant to stress. In bacteria, different patterns of sensitivity to various classes of toxicants have been observed between species (Bringman and Kuhn, 1980; Qureshi et al., 1982), and diverse results have been obtained for a given toxicant within a single species (Kenaga, 1978). Consequently, Dutka and Kwan (1981) concluded that if single species tests were to be used, a battery of such tests, involving at least 2-3 genera and 2-4 species, should be employed.

Mixed cultures of bacteria, which have been isolated from water (Vives-Rego et al., 1986) and sediment (Mowat, 1976; Liu, 1981), have also been used in bioassays. However, such cultures are typically maintained in high nutrient media quite unlike natural conditions (Jonas et al., 1984). The best predictions of contaminant impact are obtained using in situ procedures, although these are not always practical to manage because of

logistic difficulties. Thus, the best compromise appears to be that of conducting assays with natural samples maintained at conditions as close as possible to those found in nature.

Enzyme bioassays

The vital importance of enzymes at both cellular and ecosystem levels is universally recognized. Enzyme bioassays are generally more sensitive than population level (LD_{50}) tests (Liu, 1981) and are faster and more reliable than measurements of whole organism activity (Wieser and Zech, 1976).

Many techniques have been reported for the measurement of enzyme activities in bacterial systems including cultures and natural populations in water and sediment. These tests are usually colourimetric in principle and have included analyses of dehydrogenases (Nachlas et al., 1960), phosphatases (Morita and Howe, 1957), amylase, cellulase, urease (Duddridge and Wainwright, 1982), proteases (Burton and Lanza, 1985) glucosidases and acetylglucosidase (Kanazawa and Filip, 1986).

The physical nature of sediment often creates interference with colourimetric enzyme bioassays. While sediment in assay vessels may be removed by filtration or cetrifugation before analysis to reduce these interferences, most buffers used in enzyme assays have also been found to extract coloured compounds from the sediments, mostly humic and fulvic acids, which may cause interference (Tabatabai and Bremner, 1969; Duddridge and Wainwright, 1982). In order to avoid these problems, some

researchers have attempted to extract the contaminants in the sediment and use the extract in the toxicity assays. However, no extraction procedure has been developed (or is likely) that can quantitatively extract all contaminants. The most common extraction procedure is the elutriation technique developed by the U.S. Army Corps of Engineers and the U.S. Environmental Protection Agency to assess the release of contaminants during dredging and disposal activities. Elutriation involves mixing and aeration of a sediment/water mixture, and passing the supernatant through a 0.45um pore size filter (Mudroch and Davies, 1985). Unfortunately, elutriates generally are not good predictors of sediment toxicity as much better correlations exist with sediment composition than with elutriate composition (Laskowski-Hoke and Prater, 1981b). Furthermore, water extracts, such as elutriates, cannot be used to assess the impact of neutral, non-ionic compounds, such as aromatic or chlorinated hydrocarbons (Williams et al., 1986). Organic solvents such as dimethyl sulfoxide (DMSO) and hexane/dichloromethane have also been used to extract toxicants from sediments (Samolloff et al., 1983; Tabata et al., 1984; Schiewe et al., 1985). Still, the inability of a solvent to simultaneously extract different classes of contaminants remains a serious drawback. Furthermore, physical exposure to sediments appears to be important in bioassays (Laskowksi-Hoke and Prater, 1981a; Powers et al., 1982) as quantitative and qualitative differences in responses to sediments and elutriates have been reported (Peddicord and McFarland, 1976; Ewing and Nalewajko, 1987).

Much of the research carried out on the effects of pollutants on sediment processes has been done with sediments suspended in water (slurries) in ratios of 1:1 to 1:1000 (Griffiths et al., 1977; Novitsky, 1983a; Boon, 1986). This simplifies the addition of experimental substrates during the assay procedure and ensures their even distribution throughout the sediment. However, due to the disruption of sediment structure, what is measured is not natural rates of activity but responses to alterations in nutrient and substrate conditions. While any sampling method will upset the physical structure and alter the microgradients of natural sediments, taking sediment cores appears to be the least disruptive. Metabolic activity has been shown to be roughly one order of magnitude lower in intact cores than in slurries (Novitsky, 1983b; Meyer-Reil, 1984). Meyer-Reil (1986) suggested that this phenomenon was the result of bacteria in slurries being exposed to higher nutrient concentrations since microclimates were disrupted and bacteria evenly distributed within the reaction vessel. Meyer-Reil (1978) and Novitsky (1983b) placed dilute substrates over intact sediment cores and used low vacuum to draw the substrate into the cores. This method involves replacement of the interstitial water and tends to disrupt microgradients within the sediment. A far superior technique is injection of small aliquots of test substrates (10uL) into intact cores (Meyer-Reil, 1986; Ansbaek and Blackburn, 1980). The small volume of introduced substrate prevents significant alteration of the amount of interstitial water in the sediment and injection at short intervals (0.5 to 2.0 cm) down the core ensures reasonably even distribution of the substrate.

NEW APPROACHES TO MICROBIAL BIOASSAYS IN SEDIMENTS

Enzymes - fluorogenic substrates

Recently, many of the disadvantages inherent in colourimetric enzyme assays have been eliminated with the use fluorogenic substrates. Most recently, fluorescent 4-methylumbelliferone (MUF) substrates have become popular and have been used to measure phosphatase (Petterson and Jansson, 1978) glucosidase (Hoppe, 1983; Somville, 1984), protease (Somville and Billen, 1983) and chitinase (O'Brien and Colwell, 1987). These substrates are highly enzyme specific and are not easily broken down by simple chemical processes commonly found in sediments (Hoppe, 1983). The high sensitivity of fluorogenic substrates allows much shorter incubation times than chromogenic substrates and measurement of low activity (Petterson and Jansson, 1978; Somville and Billen, 1983; Somville, 1984). Fluorogenic substrates are generally used at concentrations three orders of magnitude lower than their chromogenic counterparts (Perry, 1972), and as a result, enzyme activities can be measured at their ambient, not potential, rates (Meyer-Reil, 1986).

Bacteria are unable to take up high molecular weight compounds directly and must secrete enzymes to degrade these compounds to low molecular weight substances which are then easily assimilated. These exoenzymes are vital to microbial metabolism in sediments and represent an excellent opportunity for the measurement of toxicant impacts. The potential use of one of these exoenzymes, B-glucosidase, as a bioassay tool was tested

using a fluorogenic substrate. B-glucosidase breaks down cellobiose, the repeating disaccharide unit that forms cellulose. In aquatic sediments, cellulose, from plant material, forms a major part of the organic carbon pool. Thus, inhibition of glucosidase activity would have a detrimental effect on aquatic ecosystems.

Preliminary experiments conducted in our laboratory with sediment slurries using the fluorogenic substrate 4-methylumbelliferyl glucoside were conducted with sandy sediments collected at the mouth of Halifax Harbour, to determine the sensitivity of B-glucosidase to heavy metals. B-glucosidase activity was observed within 30 minutes and remained linear for a two hour period (Figure 1). The low activity in formalin fixed samples suggests that the extracellular enzyme activity in the sediment slurries was very low and that the increased activity, upon substrate addition, is the result of substrate induced enzyme synthesis. Thus, B-glucosidase activity can be used as a quantitative measure of microbial metabolism.

B-glucosidase activity was measured over a range of metal concentrations in order to calculate EC_{50} values. In response to six metals surveyed, B-glucosidase activity typically decreased in a linear fashion with the logarithm of the metal concentration (e.g. Figure 2). The low variance between replicate measurements was typical for all experiments, showing excellent reproducibility. The EC_{50} values calculated from this data (Table 1), are roughly within the same order of magnitude as those reported by Barnhart and Vestal (1983) measured with a

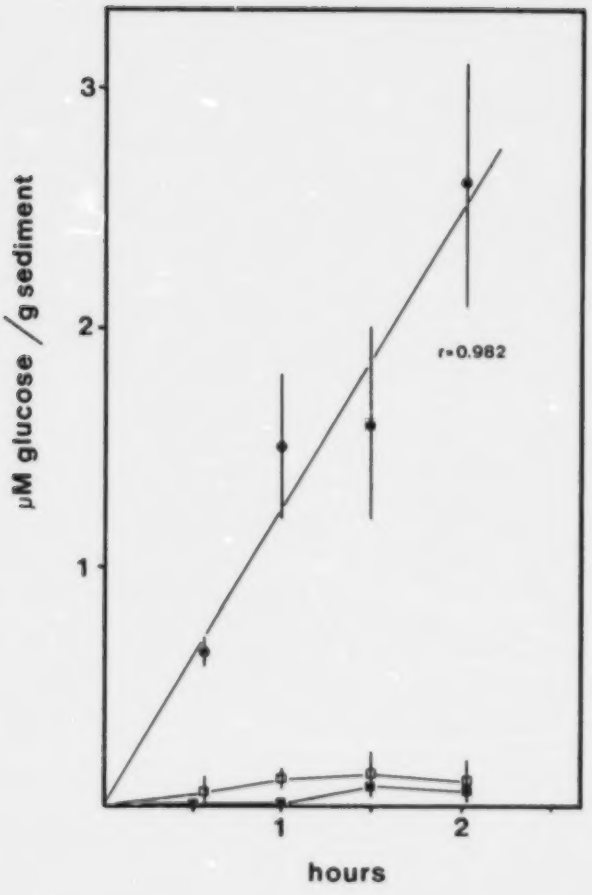

Figure 1: Time course of B-glucosidase activity in live (●), autoclaved (■) and formalin treated (□) marine sediments.

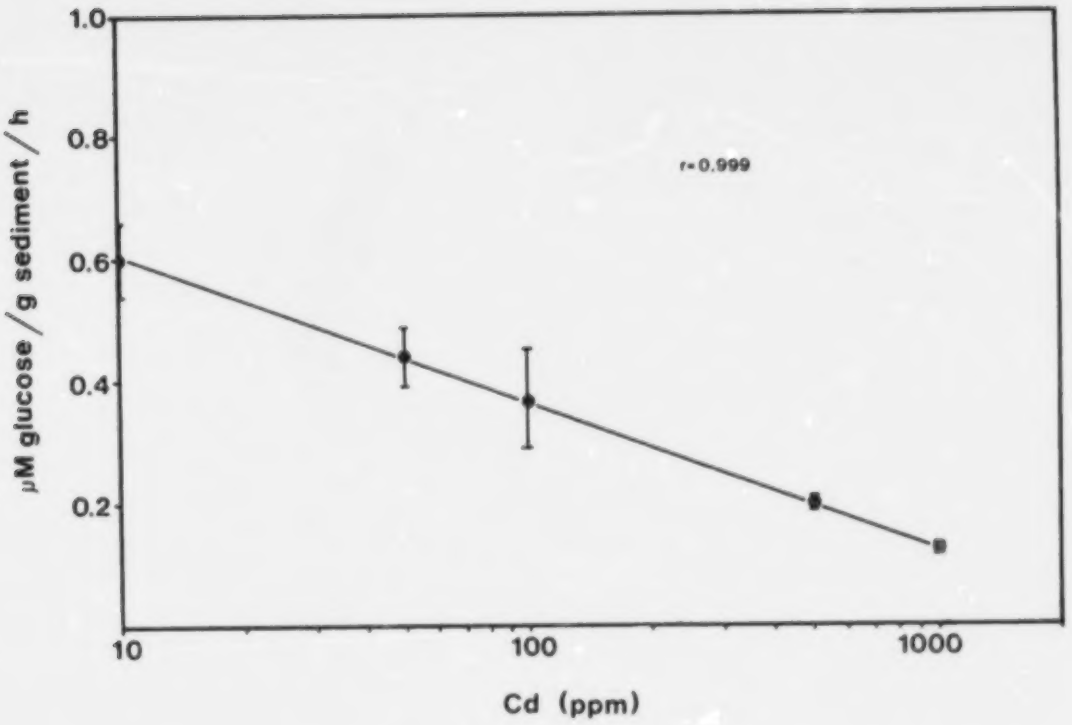

Figure 2: Effect of cadmium on B-glucosidase activity in marine sediment.

Table #1: Toxicity of Heavy Metals to B-Glucosidase activity in Aquatic Sediments.

Metal	EC_{50} (mg L^{-1})	
Hg	27.2[a]	26[b]
Zn	28.8	100
Cu	40.9	10
Cd	71.9	100
Pb	129	-
Ni	130	60

a: This study
b: Barnhart and Vestal (1983)

colourimetric substrate. However, with the fluorogenic substrate, only a 1 hour incubation period was required, a period much less than that used in colourimetric assays. The differences between our study and that of Barnhart and Vestal (1983) are most likely attributed to differences in the sediments used. These preliminary results show, nevertheless, that B-glucosidase is sensitive to heavy metal contaminants, and that this can easily be quantified with fluorogenic substrates. Glucosidase activity has been shown to be well correlated with the incorporation of organic substrates including glucose, thymidine and amino acids (Somville, 1984; Somville and Billen, 1983). The potential of other fluorogenic substrates in exoenzyme bioassays is now under investigation.

Radiotracers - thymidine incorporation

The uptake of radiolabelled substrates, including glucose, glutamate, aspartate, lactate, acetate, urea and thymidine, has been used to assess microbial activity in the aquatic environment (Litchfield et al., 1979; Novitsky and Kepkay, 1981; Fuhrman and Azam, 1982; Novitsky, 1983a, 1983b; Lee et al., 1985). Toxic metals and the water soluble fraction of crude oil have been shown to depress the intracellular incorporation of ^{14}C-glucose and ^{14}C-glutamate (Albright et al., 1972; Mills and Colwell, 1977; Henson and Hayasaka, 1982). In terms of organic uptake kinetics, the response by microorganisms to toxicants is typically an increase in turnover time and a decrease in the maximum uptake rate (Albright et al., 1972). However, the

effects of a pollutant on the uptake of one substrate cannot be used directly to predict the effects on the uptake of another substrate (Griffiths et al., 1977).

Recent experiments have shown that thymidine is taken up by nearly all marine bacteria, and its rate of incorporation into DNA is well correlated with growth (Pollard and Moriarty, 1984). Thymidine incorporation into DNA has been shown to be more sensitive to toxicants than uptake of other organic substrates such as glucose and glutamate since cellular growth is affected by a wider range of metabolic processes (Jonas et al., 1984). Furthermore, in regards to sediment bioassays, ^{3}H-thymidine incorporation into DNA appears to be an ideal process to monitor since it does not respond quickly to physical disturbances of sediment integrity (Bell and Ahlgren, 1987). The debate about isotopic dilution of the intracellular thymidine pool (Pollard and Moriarty, 1984; Riemann et al., 1987; Fuhrman and Azam, 1982; Findlay et al., 1984) is only of minor significance in bioassay studies. In pollutant impact assessments, toxicant effects can be assessed on relative rather than absolute growth rates by measuring thymidine incorporation in treated samples relative to an "uncontaminated control" (e.g. Figure 3).

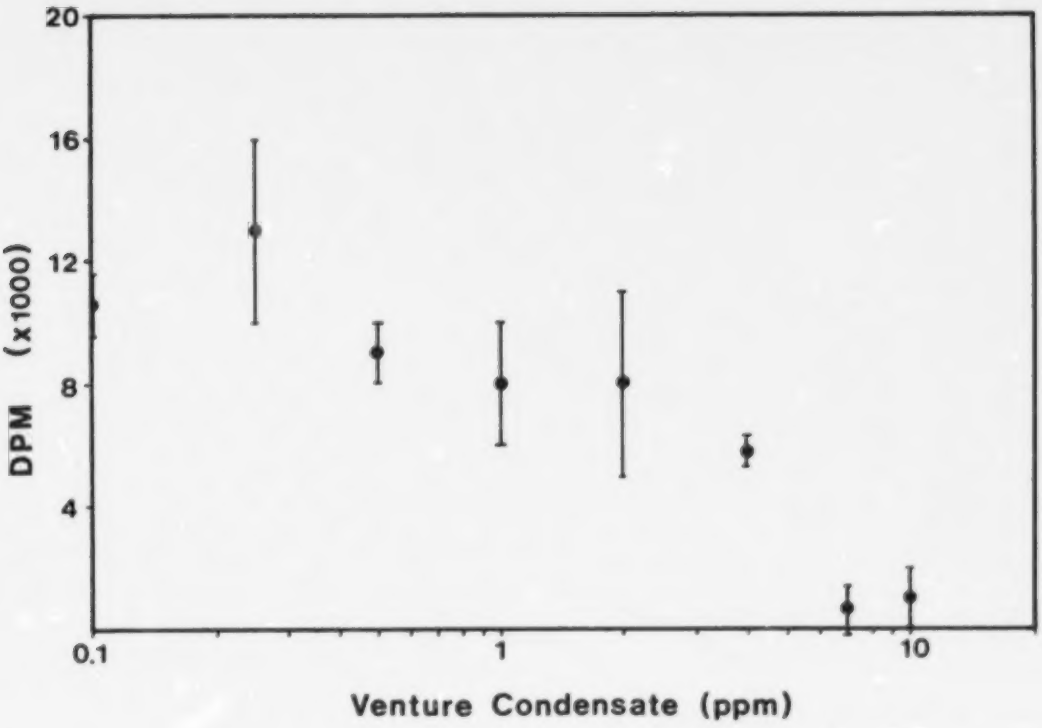

Figure 3: Effect of the water soluble fraction of Venture condensate on ^{3}H-thymidine incorporation in marine sediment.

SUMMARY

Simple and cost-effective tests are required to estimate the toxicity of waste materials to be disposed of in the sea. Bacterial bioassays can be used to monitor conditions at dumpsites before and after waste disposal events and to evaluate wastes to determine suitability for oceanic disposal. The major factor, however, concerns the ecological validity of toxicity assessment procedures. In order to gain complete understanding of the environmental effects of contaminants, their effects at the ecosystem level, not just the population and community level must be examined. Furthermore, the focus should be on basic processes, such as heterotrophy and autotrophy, since a disturbance in these will have ramifications throughout the ecosystem (Ausmus, 1984).

In terms of environmental impact assessment, single species tests, like *in vitro* enzyme inhibition tests, bear little resemblance to natural mixed assemblages. Thus it is important to use natural assemblages for bioassay tests, if results are to be extrapolated to nature with any reliability. The traditional bioassay tests using bacteria, based on the measurement of growth rates, oxygen consumption, or colony formation, required lengthy incubations (>24 hrs.), which resulted in a single concentration of toxicant being exposed to increasing biomass. These long periods of incubation also increased the risk of selecting resistant forms and thus, the effects of a toxicant on these forms, not the whole population, were measured. If reliable estimates of disposal impacts are to be achieved by the

laboratory tests, the responses of sediment microbiota to pollutants must be measured in intact sediment cores, to minimize the disturbance of fine sediment structure. Under these experimental conditions, the use of radiolabelled organic substrates (especially ^{3}H-thymidine) and fluorogenic enzyme substrates appears to offer the most promise for toxicity testing of waste materials.

ACKNOWLEDGMENTS

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ENVIRONMENTAL ACCEPTABILITY OF OCEAN INCINERATION EMISSIONS AND SOME REMARKS ON FUTURE RESEARCH AND REGULATIONS

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Our knowledge about incineration ship emissions and their environmental acceptability is reviewed critically and put into perspective. Some suggestions for possible further chemical and ecotoxicological research are made.

In order to remove the last doubts on the environmental acceptability of incineration ship emissions, all that is needed is more knowledge about the emission of persistent semi-volatile organic substances, surviving destruction or newly formed during combustion.

The analysis for these substances should be carried out with a higher sensitivity than has been done up till now. New developments in stack sampling will be discussed.

Toxicological testing must concentrate on these substances and avoid interferences by other components of the stack gases. Ecotoxicological tests in the field should primarily be carried out only when the results of model calculations, based on emission data obtained with more sensitive methods and preferably not based on analytical detection limits, prove them meaningful.

Some remarks will be made on the (non)relation between DE and CE. It will be shown that the DE concept has serious limitations when it comes to protection of the marine environment. The same holds for the POHC concept as presently defined. Suggestions for a revision of these concepts are made.

In periodic test burns with a standard waste type, testing for selected POHC's is adequate. In a test burn with a new type of incinerator and/or waste, a thorough analysis for semi-volatile organic pollutant (SVOP) emissions should be carried out.

An international consensus on the appropriate way of carrying out test burns is recommendable.

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ASSESSING POTENTIAL EFFECTS OF INCINERATING HAZARDOUS WASTES AT SEA: USING RESEARCH TO ANSWER MANAGEMENT QUESTIONS

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ABSTRACT

In 1985, the U.S. Environmental Protection Agency (EPA) developed a strategy to address the research necessary for accurately measuring possible environmental and public health effects of incinerating hazardous wastes at sea. Although existing data indicated that effects would be minimal, the research strategy was designed to ensure that possible effects would be adequately and accurately assessed.

These methods have been developed and evaluated in a series of land-based tests, and they have been found suitable for use in research to be conducted at sea. Plans for conducting research at sea will include emissions sampling and conduct of toxicity tests in conjunction with monitoring for emissions constituents in the air, surface water, sea-surface microlayer, and organisms that live in surface waters (neuston). Samples from the sea-surface microlayer will also be used as test media for toxicity tests. Results of the research conducted at sea may be used to refine the procedures to be used in permitting and monitoring ocean incineration activities.

This paper describes development of EPA's ocean incineration regulation and an operating program to administer the regulation and EPA's research strategy. We also discuss the use of those research results in determining conditions for ocean incineration permits and in developing short- and long-term monitoring programs.

BACKGROUND

With the passage of the Resource Conservation and Recovery Act (RCRA) in 1972, and recent amendments in 1974 and 1980, the U.S. Environmental Protection Agency (EPA) has more actively encouraged reduction and recycling of wastes. EPA has also explored alternatives to land-based waste disposal options. One of these options is destruction of wastes through high temperature incineration at sea. EPA, under the authority of the Marine Protection, Research, and Sanctuaries Act of 1976 (MPRSA), is the United States agency charged with regulating the disposal of waste materials in the ocean. The destruction of wastes through ocean incineration is one of the disposal activities regulated under the MPRSA.

The technology for incinerating hazardous wastes at sea was developed in Europe nearly 20 years ago. In the United States, ocean incineration is not currently a commercially available option for disposal of hazardous wastes. EPA has, however, been evaluating this technology for 13 years. Within EPA's Office of Water, the Office of Marine and Estuarine Protection is responsible for regulating ocean incineration.

Although the MPRSA gives EPA the authority to regulate ocean incineration, the actual procedures, guidelines, and requirements for regulating ocean incineration were initially in the Ocean Dumping Regulations (40 CFR Part 228). Between 1974 and 1982, research permits for

incineration were granted by EPA under the Ocean Dumping Regulations for a series of four hazardous waste burns. The objectives of these burns included testing the efficiencies of the incinerators for destroying the hazardous materials and to evaluate the operating conditions of the incinerator vessels.

Three of the four series of burns of organochlorine wastes, including PCBs for one burn, were carried out in the Gulf of Mexico aboard the M/T Vulcanus I in 1974-45, 1977, and 1981-82. The fourth, also conducted aboard the M/T Vulcanus I, was conducted in 1977 in the South Pacific Ocean, near the Johnston Atoll. Herbicide Orange was burned at that time. In addition, EPA monitored a European burn of organochlorine wastes on the M/T Vulcanus II in 1983.

Following the 1982 burn of PCB waste in the Gulf of Mexico and the 1983 burn of organochlorine wastes in the North Sea, EPA received an application from Chemical Waste Management, Inc. (CWM) for special (permits to operate commercially) and research permits. CWM proposed to burn PCB and DDT wastes. Upon evaluation of public comment on the proposed permit, EPA denied the permits to CWM (U.S. EPA, 1984). In denying the permits, the EPA Assistant Administrator for Water directed EPA to initiate development of specific ocean incineration regulations and to design a comprehensive research plan for more accurately and completely examining the potential environmental and public health risks of incinerating hazardous wastes at sea. The new regulations would be more relevant to the incineration

process than were the Ocean Dumping Regulations. Although existing information had indicated that effects would be minimal, the research plan would describe how work conducted to date and new information would be used to develop a greater understanding of the ocean incineration process.

OCEAN INCINERATION REGULATION

In February 1985, EPA issued the Proposed Ocean Incineration Regulation (50 FR 8222). Site designation and management procedures in this rule were modeled after those in the Ocean Dumping Regulations and the London Dumping Convention (LDC). Incinerator performance requirements and hazardous waste handling provisions were adapted from RCRA and the Toxic Substances Control Act. Major provisions of the regulation included the requirement to demonstrate need for ocean incineration, procedures for site selection and designation, procedures for permitting, and specifications for incinerator operations and site monitoring. The proposed regulation also included information on incinerator performance, waste specifications and limitations, and procedures for calculating allowable concentrations of metals in the waste.

During the public comment period in 1985, EPA received many comments on various aspects of the regulation. To ensure that concerns of the public are fully addressed in the Final Ocean Incineration Regulation, EPA will reopen the

public comment period on some major issues. Therefore, in late 1987, the comment period will be reopened on the following issues: determination of need for ocean incineration; financial responsibility of the applicant; concentration limits for trace metals in the waste mixture; implementation of environmental performance standards; protocols for waste sampling and analysis; port siting; permit applicant integrity; and permit application fees.

DEVELOPMENT OF AN OCEAN INCINERATION OPERATING PROGRAM

To manage the program that will result from the Final Ocean Incineration Regulation, EPA is developing an operating program that encompasses the site designation process, ocean incineration permitting, and management of sites and permits.

EPA's site designation program currently involves preparation of specific guidance that will serve as a companion to the regulation. The guidance will include a description of procedures for selecting sites and preparing an Environmental Impact Statement (or a supplement to an existing one).

The site designation process has also included evaluation of currently designated or proposed incineration sites. Since 1983, EPA has conducted several baseline surveys to existing or proposed sites as well as to areas that may in the future be appropriate for ocean incineration

sites. Data collection during these surveys has included evaluation of air and water for background levels of organic compounds of the type that may be incinerated of may result from the incineration process; characterization of the neuston (surface water) community; and quantification of seabirds and endangered species of whales and turtles.

The permitting process provided in the Ocean Incineration Regulation will be protective of the environment and will provide for public comment and participation. EPA is preparing guidance on the overall permitting process and on several specific issues, such as the preparation of contingency plans and for conducting trial burns. Contingency plans, which will be required as part of the permit application, must detail the procedures that applicants will take to avoid accidents and those that will be taken in the event of accident or unplanned occurrence. Trial burns, which will be performed prior to granting an operating permit, are designed for the applicant to demonstrate that the incinerators are capable of operating at the conditions and achieving the destruction efficiency specified by the regulation.

In designating sites and issuing permits, EPA believes that criteria developed for both site designation and for permitting will adequately protect the marine environment. EPA's monitoring programs are designed to test this assumption. EPA is currently developing monitoring programs for ocean incineration that will include assessment of compliance with permit conditions and assessment of potential environmental impacts from incineration of wastes.

EPA'S OCEAN INCINERATION RESEARCH STRATEGY

EPA's comprehensive strategy to obtain information necessary for effective management of ocean incineration activities was drafted in 1984-85 (EPA, 1985). The major focus of the research strategy was the development of a rational, scientifically defensible methodology for an updated assessment of environmental risks associated with ocean incineration. There is a significant difference between assessing risk associated with effects of a specific hazardous compound and that associated with a complex mixture, such as the incinerator emissions of a hazardous waste incinerator. Incinerator emissions are difficult to sample at high temperatures, may exist in two phases (emissions gases and scrubber water), and are not easily characterized.

The research strategy developed by EPA is divided into three major areas, each having specific objectives. The research in Study Area 1 involves development of methods for sampling incinerator emissions for chemical characterization and toxicity testing. Testing the toxicity of the emissions to aquatic organisms has involved developing a system to sample a known volume of emissions from an incinerator stack and incorporating them into seawater. The emissions-in-seawater mixture can then be used as a test medium in toxicity tests that are conducted at sea.

Study Area 2 of the research strategy involves determining the potential environmental impacts of incinerator emissions on all components of the environment and includes the use of the methods developed under Study Area 1 as well as other sampling and analysis methodology in the field. Study Area 2 is expected to culminate in the conduct of a research burn or burns at sea. Monitoring of the burn will include emissions sampling and conduct of toxicity tests in conjunction with monitoring for emissions constituents in the air, surface water, sea-surface microlayer, and neuston. Samples from the sea-surface microlayer will also be used as test media for toxicity tests. Models to be used in the permitting process will also be tested.

Under Study Area 3, detailed studies are being conducted to evaluate the impacts of ocean incineration on public health and welfare. This research area includes detailed studies of transport and fate of emissions in the air and water column, toxicity tests using organisms indigenous to potential burn sites, studies of the air-sea interface, and long-range studies to evaluate the effects of bioaccumulation of emissions constituents.

EPA has been implementing the ocean incineration research strategy since it was adopted in 1985. The results of the research being conducted in Study Areas 1, 2, and 3 will be used in developing, refining, or implementing the ocean incineration operating program. Specifically, results of the research will be used to refine permit conditions for

incinerator operations and performance monitoring and to refine the monitoring programs currently under development. Results of the research will be made available to the public when final reports have been prepared and reviewed by the scientific community.

REFERENCES

U.S. Environmental Protection Agency (1985) Incineration-at-sea research strategy. Washington, DC.

ASSESSING POTENTIAL EFFECTS OF INCINERATING HAZARDOUS WASTES AT SEA: DEVELOPMENT AND FIELD-TESTING OF THE MARINE BIOLOGICAL ASSESSMENT SAMPLER

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ABSTRACT

In 1985, the U.S. Environmental Protection Agency (EPA) developed a strategy to address the research necessary for accurately measuring possible environmental and public health effects of incinerating hazardous wastes at sea. One area of this research strategy addressed development of a method to sample incinerator emissions and introduce them into seawater for use as test media in toxicity tests. In response to the research strategy, EPA has developed the Marine Incineration Biological Assessment Sampling (MIBAS) system.

The MIBAS system samples incineration flue gas, cools the emissions, and collects them in seawater-filled impingers. Particulate matter, semivolatile organic species, and nonvolatile organic species are collected by the train. The sample taken for use in toxicity tests includes the contents of the impingers plus the contents of the probe, a flexible Teflon line, and a condenser coil.

The system uses no materials that could in themselves prove toxic to marine organisms. A recent modification of the train has provided for collection of emissions in the first impinger without bubbling, mimicking the situation in nature whereby emissions would settle onto the surface of the ocean.

The MIBAS system has been evaluated through a series of tests conducted at the Battelle Multifuel Furnace in Columbus, Ohio and the EPA Combustion Research Facility in

Jefferson, Arkansas. These tests have included spike-recovery experiments using a gas-phase spiking system to spike compounds into the emissions and then measuring them in the components of the MIBAS train.

INTRODUCTION

As part of the U.S. Environmental Protection Agency's (EPA) consideration of ocean incineration of hazardous wastes as a waste disposal option, EPA has formulated an overall research strategy (EPA, 1985). The major focus of the strategy is the development of a rational, scientifically defensible methodology of an updated assessment of environmental and public health risks associated with ocean incineration. One area of research, designated as Study Area 1, involves development of methods for sampling emissions for chemical characterization and for use in toxicity testing. There is a significant difference between assessing risk associated with effects of a specific hazardous compound and that associated with a complex mixture, such as the incinerator emissions of a hazardous waste incinerator. Incinerator emissions are difficult to sample at high temperatures, may exist in two phases (emissions gases and scrubber water), and are not easily characterized. This paper describes the method developed to sample stack emissions and introduce them to seawater for use in toxicity tests.

ALTERNATIVES CONSIDERED AND SAMPLING SYSTEM SELECTED

Because the purpose of the sampling system was to provide samples for use as test media in aquatic toxicity tests, the system had to be designed to produce samples that would simulate ocean water after exposure to incinerator emissions. It could not introduce materials that would in themselves prove hazardous to marine organisms, and it had to be capable of collecting appropriate volumes of emissions to be able to test realistic concentrations of emissions constituents.

The system had to be rugged enough to be used at sea and made of readily available materials so that repairs could be made in the field. It had to be resistant to corrosion so that it could be used at sea, aboard an incinerator vessel. Because collection of emissions samples also results in collection of hydrochloric acid (HCl) and water vapor, the system had to be able to accommodate the extra volume of material that would be collected.

One possible design for the system was a packed-bed demister train. This system uses a countercurrent, cascade scrubbing procedure to collect particulate emissions. Gaseous emissions are passed through a seawater tank, and residual water is removed in a silica gel impinger. This type of sampling system was rejected from consideration, because the technology was untested, and the ability to achieve intimate mixing between gas and liquid phases was uncertain.

Another system that was considered was a large liquid tank with dispersed bubble sparging. In this system, gaseous emissions are initially bubbled through ocean water contained in a large tank. A second tank collects any overflow from the first tank. The second tank is followed by a silica gel trap, a dry gas meter, and a vacuum pump. This system was not used, because the design was untested in stack gas sampling, and because it suffers from having only a single gas-liquid exposure.

Cryogenic sampling techniques were excluded, because the thermal fluctuations involved in freezing and thawing could modify many of the chemicals potentially present in incinerator emissions. Resin sampling techniques were also excluded, because they require either heat, which could decompose compounds of interest, or solvent flushing, which would render any sample unsuitable for use in toxicity tests.

The sampling system selected was based on the EPA Source Assessment Sampling System (SASS) impinger train, modified for direct seawater impingement and condensation. The modified system was renamed the Marine Biological Assessment Sampler (MIBAS). Because the system is a modification of an existing system, ruggedness, corrosion protection, availability of materials, and ease of transportation have already been demonstrated. The train did, however, require testing for completeness of collection in a seawater matrix.

DESCRIPTION OF THE MIBAS TRAIN

The MIBAS train, as originally configured, includes a glass-lined, water-cooled probe, flexible Teflon sample line, a water condensation coil, three glass impingers containing seawater, an XAD-2 adsorbent trap, a glass impinger containing a sodium hydroxide solution and acid indicator, an impinger with activated silica gel, a sampling pump, and a dry gas meter. The glass impingers fit into an ice chest, allowing ease in handling and a continually cooled environment. The impingers, condenser, and sample line are held together by ball-and-socket joints, clamped with knurled nuts. Good sealing of the joints is attained without the use of stopcock grease.

The MIBAS train collects particulate matter, semivolatile organic species, and nonvolatile organic species. Cooled gas (<20°C) enters the first seawater-filled impinger through a tube with four openings. These openings allow direct impingement of gas and particulate material. The second impinger is a open tube. Any particulate material should be collected in these two impingers. The third impinger uses a coarse frit on the impinger tube to improve gas-seawater contact.

Recently, an additional impinger was added to the front of the MIBAS train. To better mimic the settling and subsequent mixing of emissions constituents with seawater in the field, this impinger has a shortened stem that allows the sample to be collected without bubbling.

The resulting sample for use in toxicity tests includes the contents of the emissions sampling probe, the Teflon-lined coil, the condensing coil, and the glass impingers, as well as seawater rinses of these components. The XAD-2 resin is present to collect any organic analytes that may pass through the seawater impingers. Solvent rinses of the train are also used to measure any organic species adhering to the sampling system.

VALIDATION OF THE MIBAS TRAIN

Validation of the MIBAS train has included experiments using simulated flue gas, fuel oil burns, and hazardous waste burns at Battelle's Multifuel Furnace in Columbus, Ohio and at the EPA Combustion Research Facility in Jefferson, Arkansas.

The simulated flue gas tests were conducted to determine performance and limitations of the MIBAS train. Tests conducted have included seawater blanks, HCl addition tests, time trials, and precision tests. Seawater blanks were used to determine whether the system was subject to unacceptable contamination in the field. Simulated flue gas spiked with HCl was used to demonstrate the impact of the highly acidic environment on collection by the train. Time trials were used to determine an optimum sampling time for collection of a sample that would be of appropriate concentration for use in toxicity test. Precision tests were conducted to define the limits for the system.

Tests of the system using fuel oil were conducted as a system "shakedown," to determine whether the system could be used in a field situation such as would be encountered at sea. Toxicity tests were conducted in a mobile toxicology laboratory during the same period.

Validations of the system in a series of waste burns are being conducted to further challenge of the test protocols and to provide a "range-finding" estimation of concentrations that affect test organisms. These tests have included a series of experiments in which known chemical compounds are spiked into the gas phase of the system during sampling and into the seawater impingers before and after sampling. Recovery of these compounds is used to judge trapping efficiency of the train.

Although validation of the MIBAS train is not complete, it appears that the train is a valuable tool for collecting emissions and using them in tests to assess the potential risks associated with incineration of wastes at sea.

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PROPOSED BIOLOGICAL TESTING METHODS FOR THE UNITED STATES INCINERATION-AT-SEA RESEARCH PROGRAM

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Abstract

Ship-board toxicity tests of incinerator emissions generated during a chlorinated waste research burn will be conducted as part of the United States Environmental Protection Agency's (EPA) Incineration-at-Sea research program. The five rapid chronic tests to be used employ the inland silverside, Menidia beryllina, the mysid Mysidopsis bahia, the red macroalga Champia parvula, the archiannelid Dinophilus gyrociliatus, and gametes from the sea urchin Arbacia punctulata as test organisms. The duration of the individual tests range from two hours to seven days. The endpoints include survival, growth and reproductive effects. To date there have been three land-based trial burns to validate the sampling and test methods. They have demonstrated that the proposed methodologies can be used to test the toxicity of gaseous emissions, and that there appears to be no measurable toxicity associated with the combustion of a carrier fuel oil.

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ERL-Narragansett contribution No. 871.

Introduction

The use of at-sea incineration of wastes as a mechanism of hazardous waste disposal was developed nearly two decades ago. Since that time, European countries have routinely burned wastes aboard incinerator ships in the North Sea. The United States is presently considering this technology as an addition to its hazardous waste disposal program.

Since 1972 the United States Environmental Protection Agency (EPA) has conducted several research burns in the Pacific Ocean and the Gulf of Mexico. The EPA is currently developing a work plan for conducting future at-sea research burns. Unlike previous research burns, the proposed burn will include on-board toxicity testing of the stack emissions during the burn, in addition to numerous other biological and chemical analyses.

This paper briefly describes the biological tests that will be employed to assess the toxicity of stack emissions generated during the proposed research burn.

Proposed Methods

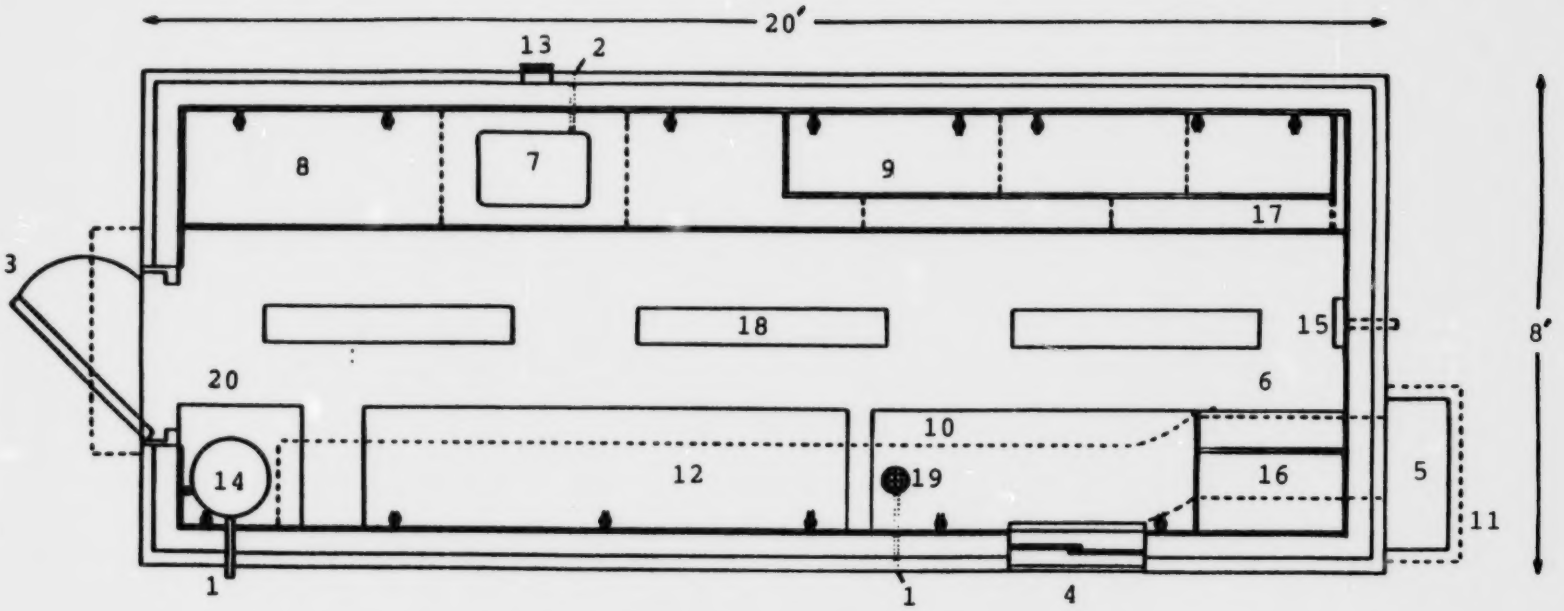
A suite of five toxicity tests has been chosen for use by the Incineration-at-Sea program. They are "rapid chronic" tests using the inland silverside, Menidia beryllina, the mysid Mysidopsis bahia, the red macroalga Champia parvula, the archiannelid Dinophilus gyrociliatus, and gametes from the sea urchin Arbacia punctulata as test organisms. The duration of the tests range from two hours to seven days. The endpoints include growth, survival and reproductive effects.

Criteria used to evaluate the suitability of the above tests included:

- Adaptability of the protocol to alternative bioassay system designs such as those used for on-site testing in mobile laboratories,
- The year-round availability of the test species and appropriate life stages,
- Bioassay endpoints that can be obtained relatively quickly (seven days or less) and are readily interpretable in terms of potential deleterious marine populations or community responses,
- Ease with which the test organisms can be cultured in the laboratory and transported to the mobile laboratory site,
- High sensitivity of the test species to the types of toxicants expected in the emissions of hazardous waste incineration, and
- Sufficient data bases available to attest to the reliability and reproducibility of the protocol.

With the exception of the Dinophilus test, all tests were developed or modified at the EPA's Environmental Research Laboratory in Narragansett, Rhode Island (ERL-N). They were developed for the Marine Effluent Testing Program to test industrial and municipal effluents discharged into the marine and estuarine environments. The Dinophilus test was developed by researchers at Battelle Ocean Sciences in Duxbury, Massachusetts.

All tests will be conducted in a mobile laboratory on-board the incinerator ship during the research burn. This lab was designed by EPA specifically for this program. It is a 20 by 8 by 8 foot modified marine shipping container (Figure 1) designed to be easily placed on the incinerator ship. The lab is equipped with temperature-controlled water baths, hot and cold



1. 3/4" Drain through sidewall w/ garden hose type fitting
2. 1 1/2" Drain through sidewall w/ garden hose type fitting
3. Exit door
4. Window - escape type
5. Board 20 220v 60 cycle
6. Refrigerator
7. S.S. sink
8. Formica cabinet
9. Hanging cabinet
10. Kneehole desk w/ counter top
11. Hood
12. 3-tiered temperature and light control system
13. 8" Exhaust fan
14. Hot Water Heater, 36 gal.
15. Breaker panel
16. Hanging cabinet w/ shelves
17. Formica cabinet w/ 5 drawers, 24" door w/ touch latches
18. Fluorescent lights
19. 4" Floor drain
20. Formica cabinet

Figure 1. Description of the mobile bioassay laboratory.

fresh water and all equipment and supplies needed to perform the five tests.

Sampling methods

Since the biological assays were designed to test the toxicity of effluents or compounds in sea water, it will be necessary to effectively transfer the constituents of the stack gas to an aqueous medium for testing. To accomplish this, the Marine Incineration Biological Assessment Sampler (MIBAS) system was developed (Figure 2). Stack gasses will be cooled and then bubbled through two liters of sea water in a series of impingers for three hours. The resulting concentrated solution in the impingers along with sea water rinses of the entire system (including the stack probe) will constitute the stock solution to be tested (for a more thorough description see Redford et al., 1987). It will then be diluted with sea water to approximate "worst-case" conditions.

Since HCl is one of the breakdown products of chlorinated compounds, the pH of the MIBAS sample will be outside of the tolerence limits of the test organisms. During preliminary tests the pH ranged from 0 to 1. Therefore, the pH and salinity of the sample will be adjusted to 8.0 and 30 o/oo, respectively, before testing. Sodium hydroxide and deionized water will be used for these adjustments. This neutralization process was deemed acceptable because HCl deposited on the sea surface during an at-sea burn would be rapidly neutralized by the ocean's natural buffering system.

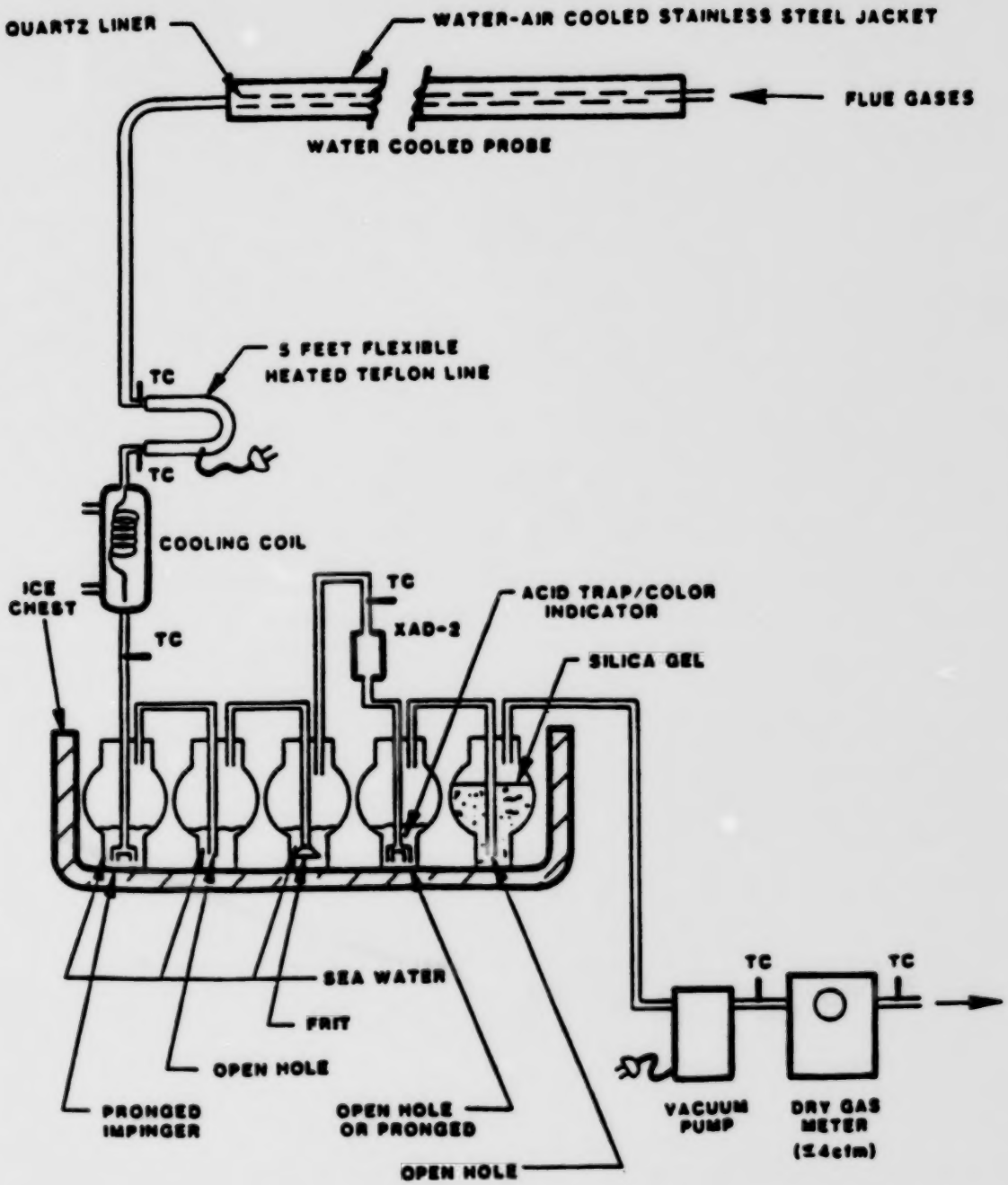


Figure 2. Description of the MIBAS system.

Menidia beryllina

The fish growth/survival test is a seven-day assay conducted under static renewal conditions (test solutions are renewed daily). Fifteen seven-day old individuals (laboratory reared) are used per replicate, with three replicates per treatment. The treatments consist of dilutions of the pH and salinity-adjusted MIBAS sample (10, 3.2, 1.0, 0.32 and 0.1 percent), a control using clean sea water collected in advance from the incineration site ("site water"), and a control using reconstituted Narragansett Bay hypersaline brine (as a reference for the site control). Site water is used for all dilutions of the MIBAS samples. Fresh MIBAS samples are obtained daily for renewals.

The test chambers are glass Mount-Nordberg chambers (rectangular chambers with a sump area for solution renewals) containing approximately one liter of test solution. The chambers are sufficiently large, heavy and tall so that they should not be disturbed by the the ship's motion.

To initiate a test, chambers containing the test animals are placed in a random array in one of the 25°C water baths. Temperature, salinity and dissolved oxygen are measured daily just prior to solution renewal. A light/dark cycle of 14/10 hours is maintained. Dead animals are removed, recorded and discarded daily. Animals are fed Artemia nauplii daily after solution renewal, with care being taken to assure a uniform amount of food per replicate. During solution renewals, excess Artemia and debris are removed.

On day 7 (the last day of the test) surviving fish are transferred by replicate to a vial containing 4% buffered formalin. Upon returning to the laboratory these fish are dried and weighed. The endpoints for this test are growth (final weight differences between control and experimental animals) and percent survival. All data are statistically analyzed by Analysis of Variance (ANOVA) and Duncan's Multiple Range test. Survival data are arcsine root transformed prior to analysis.

During the expected 19 day research burn this test will be performed twice. Additional information on this test can be found in Heber et al. (1986).

Mysidopsis bahia

The mysid assay is very similar to the fish test. This test is a seven-day static renewal assay using the same treatments described above. The test chambers are 12 ounce disposable plastic drink cups (pre-soaked in sea water) containing approximately 150 ml of test solution. There are 8 replicates per treatment with five animals per replicate.

The test is initiated with seven-day old laboratory-reared individuals. The test chambers are randomly arranged in racks in a 26°C water bath. Water quality measurements and solution renewals are performed daily. As with the fish, the food source is Artemia nauplii.

On day 7 (the last day of the test) surviving mysids are counted and removed for observation. They are observed under a dissecting microscope for maturity and egg production. The

numbers of males, immature animals, females with eggs and females without eggs are recorded. All individuals are then dried and weighed. Animals are observed live because upon preservation they turn opaque thereby making the microscopic observations impossible. Growth (as dry weight), percent survival and percent females bearing eggs are statistically analyzed by ANOVA and Duncan's Multiple Range test. Growth is determined separately for males and females. Survival and egg production data are arcsine root transformed prior to analysis.

During the expected 19 day research burn this test will be performed twice. Additional information on this test can be found in Lussier et al. (1985).

Champia parvula

The red macroalga test consists of a static two-day exposure (no solution renewal) followed by a five to seven-day recovery period. The test concentrations are identical to those described above, except the reconstituted Narragansett Bay hypersaline brine is replaced with autoclaved Narragansett Bay sea water. There are three replicates per treatment with five plants per replicate. The exposure chambers are plastic specimen cups containing 80 ml of test solution plus nutrients.

To initiate a test, five female plant tips (ca. 1 cm) and one larger male plant tip are placed in each exposure cup. The cups are covered and placed in random order on a shaker table. They are maintained at a temperature of 20°C and a light/dark cycle of 16/8 hours. At the end of the exposure period, the

male tips are discarded and the female tips placed in culture bottles containing clean site water. The bottles are placed on the shaker table and allowed to "grow out" for five to seven days.

At the end of the recovery period, the plants are observed microscopically for cystocarp production. Cystocarps are evidence of sexual reproduction and appear as bumps on the parent plant. ANOVA and Dunnett's test are used to compare the number of cystocarps per plant across treatments.

During the expected 19 day research burn this test will be conducted six times. Additional information on this test can be found in Thursby and Steele (1985).

Dinophilus gyrociliatus

The archiannelid test is a seven-day life-cycle static renewal assay. A minimum of four one to two-day old juveniles are exposed per replicate. There are five replicates per treatment. The treatments consist of dilutions of the adjusted MIBAS sample (8.3, 4.2, 2.1, 1.0 and 0.5 percent) plus a site water control. Unlike the tests described above, this test is conducted at a salinity of 25 o/oo.

To initiate a test, individuals are placed in glass Stender dishes containing 10 ml of test solution. These chambers are placed in the dark and maintained at 21°C. Due to the limited volume of test solution, dissolved oxygen and salinity are monitored on fresh and 24-hour old composite samples for each treatment on days 1 and 6 only. Temperature of the test

compartment is monitored continuously with a temperature probe.

Daily observations are made of the number of surviving individuals, number of eggs laid, number of eggs per capsule and the number of juveniles released. Animals are fed once at the start of the test with two drops of a 0.5% (w/v) suspension of 105 μm sieved spinach.

Upon termination of the test on day 7, the above observations are made and the number of eggs visible within the coelom of each worm is recorded. The endpoints for this test are survival and reproductive success. Four and seven-day LC50s are determined by the Spearman-Karber method with Abbott's correction. Reproductive success (the total number of eggs produced per surviving female) is statistically analyzed by ANOVA and Duncan's multiple range test.

During the expected 19 day research burn this test will be performed twice. Additional information on this test can be found in Carr et al. (1986).

Arbacia punctulata sperm cell test

The sea urchin sperm cell test is a very rapid test using egg fertilization as in indicator of sperm viability. The duration of the test is approximately 1 hour and 20 minutes. There are four replicates per treatment. Treatments consist of serial dilutions of the adjusted MIBAS sample and both a site water and reconstituted Narragansett Bay hypersaline brine control. This test is conducted at a salinity of 30 o/oo and a temperature of 20°C.

To initiate a test, sperm is obtained from four mature males by electrical stimulation. Fifty µl of diluted sperm from one animal is used per chamber for each replicate. The test chambers are scintillation vials containing 5 ml of test solution. Vials are incubated for one hour at 20°C. One ml of a diluted egg suspension (eggs also obtained via electrical stimulation) is then added to each vial. All vials are incubated at 20°C for an additional 20 minutes and then preserved by adding two ml of 10% buffered formalin. The eggs are then observed under a microscope for the presence of a fertilization membrane, indicating that fertilization has occurred.

The endpoint in this test is the percent fertlization per treatment relative to the control. An EC50 is determined by the Spearman-Karber method with Abbott's correction.

During the expected 19 day research burn this test will be conducted daily. Additional information on this test can be found in Nacci et al. (1985).

Preliminary Verification and Discussion

To date, there have been three land-based test burns within this program. The intent of these studies has been to validate the use of the proposed test methods for measuring the toxicity of gaseous emissions.

During the first burn, uncontaminated fuel oil was burned. To simulate the conditions encountered during the incineration of a chlorinated waste, one of the treatments consisted of the

MIBAS sample spiked with HCl and then pH adjusted. No toxicity was detected in any of the treatments by any of the assays.

PCB-contaminated waste was incinerated during the second burn. Toxicity associated with the emissions was measured by all five tests only at concentrations above EPAs estimates for "worst case". The Arbacia and Champia tests were the most sensitive of the five tests. The Dinophilus assay was the most sensitive of the three seven-day tests.

During the third test, a mixture of carbon tetrachloride and chlorobenzene was burned. Again toxicity was measured in all five tests. The effect concentrations were slightly higher than those generated during the PCB burn and the order of species sensitivity was the same.

The data from the three land-based test burns demonstrate that the proposed biological testing methodologies can be successfully used to measure the toxicity of the gaseous emissions generated during the incineration of chlorinated wastes. The lack of toxicity associated with the fuel oil burn suggests that the toxicity observed in the other burns (which also used fuel oil as a carrier for the waste) can be attributed to the combustion of the waste itself and not the carrier oil.

With the verification of these methods completed, they can now be employed to measure the toxicity of the emissions generated during an actual at-sea research burn during the next phase of this project.

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THE CHANGING CHARACTERIZATION OF OCEAN INCINERATION WASTE STREAMS AND ITS IMPACT ON THE INDUSTRY DEMAND

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Ocean incineration has been employed since 1969 for the destruction of liquid hazardous waste. Until recently this technology was used primarily for waste streams with very high chlorine and low water content and consequently could not be incinerated as effectively on land-based units.

There has been a gradual change in the characterization of typical waste streams incinerated on Vulcanus II to the current composition of lower chlorine and heat content levels and much higher water concentrations. This historical shift is the result of technological advances and acceptance of the ocean incineration process. These changes have also caused a dramatic increase in the number of generators served by this technology, a trend expected to continue.

This paper will present data demonstrating the shift in waste stream characterization and its impact on the available market. This will also serve as another factor supporting the continual use of ocean incineration.

INTRODUCTION

The ocean incineration industry originated in 1969 when the dumping of certain hazardous wastes in the North Sea became prohibited. Through an international conference, Europe regulated the disposal of highly chlorinated waste by requiring it to be incinerated, but the scrubber technology was not sufficiently advanced to incinerate the hazardous waste on land. Thus ocean incineration technology had its birth.

In 1972 Ocean Combustion Service BV, Rotterdam, The Netherlands, converted a dry cargo vessel to an incineration ship named the Vulcanus. During the next eight years, this vessel operated primarily in the North Sea but also incinerated waste from the United States in the Gulf of Mexico and the Pacific in 1974 and 1977. The United States Environmental Protection Agency (U.S. EPA) was impressed by this technology and encouraged that it be brought to the United States.

In 1980 Waste Management, Inc. (WMI)* acquired Ocean Combustion Service and the Vulcanus vessel with the intent of providing ocean incineration to U.S. generators while continuing to service the European market. The company immediately

* Although the Waste Management, Inc. (WMI) name is used throughout this paper, many of the activities were performed through the Chemical Waste Management, Inc. (CWM) subsidiary.

applied for permits to operate the ship in U.S. waters. Vulcanus continued to operate in Europe under the new ownership but returned to the U.S. in December, 1981 and again in August, 1982 and incinerated two shiploads of PCB (polychlorinated biphenyls) waste in order to demonstrate the effectiveness and safety of the process. In anticipation of receiving a U.S. permit, WMI built a new ship, the Vulcanus II (Figure 1), and launched it in November, 1982 in Europe where it has since been operating. In 1983, the Vulcanus was retrofitted to bring it to the state-of-the-art and renamed Vulcanus I. WMI planned to operate Vulcanus I in Europe and Vulcanus II in North America.

While the U.S. PCB research burns were successful and the U.S. EPA stated that no other research was necessary to prove the technology, in May, 1984 it reversed its tentative decision of November, 1983 by denying WMI research or operating permits. EPA's decision was based on the need to obtain more information before proceeding with the program. Specifically, EPA requested a market and risk assessment of land-based and ocean incineration (EPA, 1985a), specific ocean incineration regulations (proposed in February, 1985 but not yet final), and the development and implementation of a research plan to answer the questions raised by Congress, environmental groups, and the public community.

In September, 1985 WMI submitted a permit application to conduct this proposed research. EPA made a tentative de-

FIGURE 1

termination in December, 1985 to issue the research permit, but after a lengthy comment period including public hearings, EPA denied the permit in May, 1986. The Agency believed that a research burn was not necessary at that time, and it would be more appropriate to conclude the promulgation of ocean incineration regulations before proceeding with any operations.

EPA's vacillation on the entire ocean incineration issue forced WMI to file a suit against EPA in December, 1986. WMI requested the U.S. Federal Court to have EPA promulgate new regulations immediately or to process a permit application according to the existing regulations. In September, 1987 the court dismissed the suit after concluding that EPA had complied with all relevant legal requirements.

With respect to the long awaited ocean incineration regulations, they have not yet been reproposed and thus EPA missed another deadline. The EPA regulatory calendar listed the issuance of the reproposed regs for early September, 1987. The objective of the reproposal is to solicit additional comments from the public on several controversial issues which will probably include:

1. determination of need for ocean incineration
2. demonstration requirements of financial responsibility

3. appropriate metal limitations
4. appropriate waste analyses
5. appropriate environmental performance standards
6. criteria for evaluating the integrity of the applicant

Of these six (6) issues, one of the most controversial is the demonstration of need for ocean incineration. While this paper does not address directly the question of "demonstration of need", it should shed some light on the "need for ocean incineration". A critique of other major issues was presented at the Sixth International Ocean Disposal Symposium (Nassos, 1987).

Since the inception of ocean incineration, there have been several changes that have impacted this business. Changes in the customer (generator) base, the waste stream composition and the incineration technology have each had an influence on the direction and need for ocean incineration.

Changes in the Customer (Generator) Base

Although it would be a real pleasure to discuss the U.S. ocean incineration customer base, this is obviously impossible. There are, however, considerable data on the only real market today: Europe.

In 1972, when Vulcanus first went into operation, only two generators contributed all the waste which comprised the ship's cargo for that year. Almost 18,000 metric tons (MT) were incinerated that year. During the following year (1973) the number of generators increased to nine (9), and the volume incinerated more than doubled to over 36,000 MT. By the end of the decade the number of generators increased to 126 while the volume was up to only 43,000 MT. Last year (1986), 2201 different facilities generated almost 86,000 MT.

A visual representation of the changes in generator base is more meaningful and is shown in Figure 2. As can be seen, the number of generators has increased almost exponentially while the volume incinerated has had a more gradual increase with some intermittent declines. Consequently, the volume per generator has decreased significantly.

The increase in the number of generators and the volume incinerated can be explained by a combination of the following factors:

- Stricter regulations and enforcement by the authorities over waste disposal has forced many businesses to comply with the regulations and thus find more responsible disposal methods for their waste. Many small firms have selected ocean incineration due to this tighter legislation.

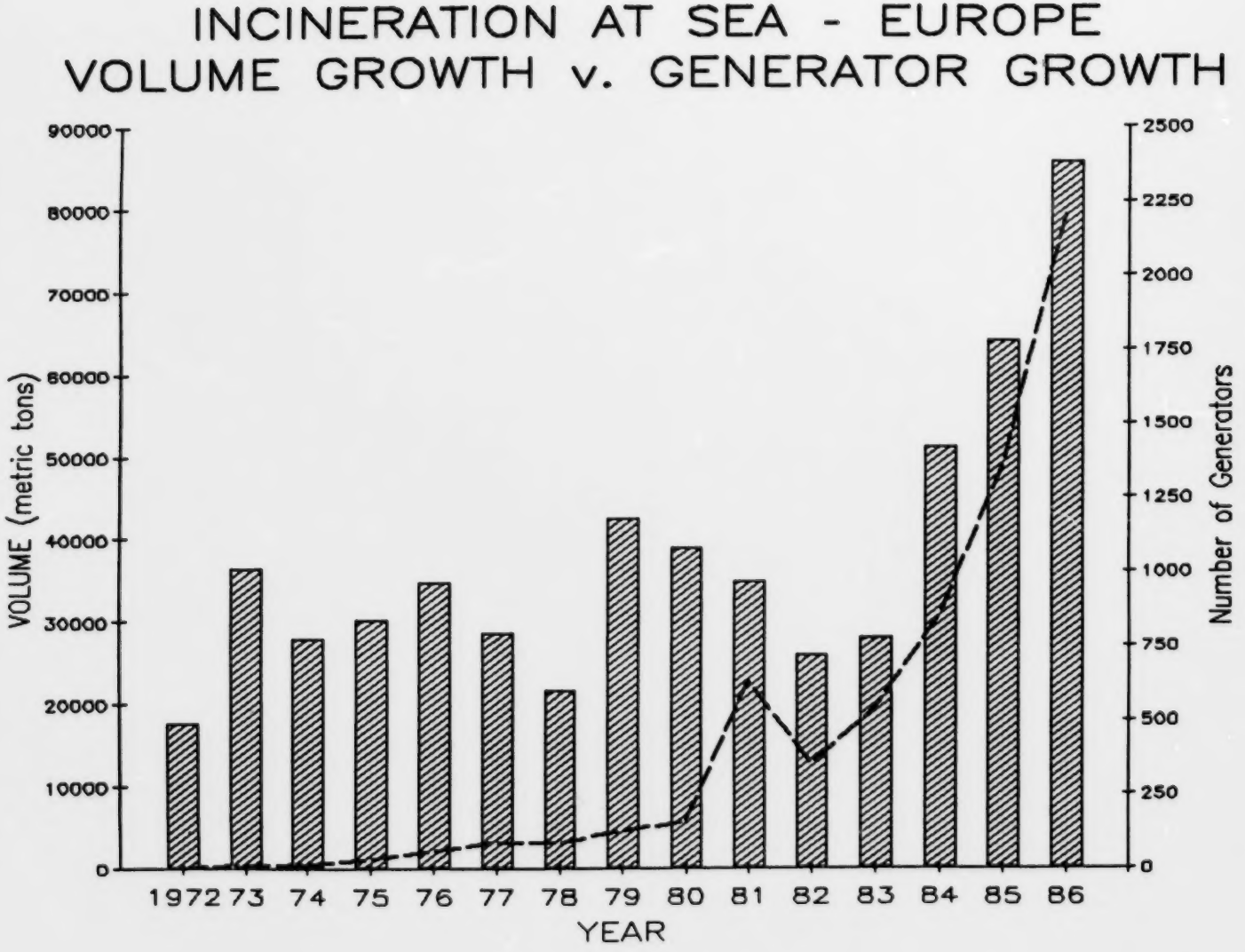
INCINERATION AT SEA - EUROPE
VOLUME GROWTH v. GENERATOR GROWTH
VOLUME (metric tons)
90000
80000
70000
60000
50000
40000
30000
20000
10000
0
Number of Generators
2500
2250
2000
1750
1500
1250
1000
750
500
250
0
1972 73 74 75 76 77 78 79 80 81 82 83 84 85 86
YEAR

FIGURE 2

- The environmentalist movement protests against generators which dispose of waste in an unsound manner. The media give attention to the events, and businesses are careful to avoid adverse publicity.

- More generators of liquid organic waste became dependent on ocean incineration as the technology achieved wider acceptability.

- Ocean incineration became the most cost effective technology for the disposal of other waste streams in addition to the highly chlorinated ones which initiated the demand for this technology.

- The incineration-at-sea technology advanced such that certain waste streams previously unacceptable are now easily incinerated.

The decrease in the quantity of waste per generator during the last few years, especially by the large generators, can be explained as follows.

- Production methods have become more efficient whereby better use is made of raw materials. Due to waste minimization programs, there has

been a corresponding reduction in the quantities of waste materials and in the occurrence of off-spec product.

- The large generators of waste reduced their dependence on ocean incineration by waste reduction and/or alternative treatment methods.

- Higher raw material prices have made the recycling and reuse of waste products more attractive.

- Economically feasible recycling has led to the formation of firms which collect small amounts of waste from many companies. Ocean Combustion Service receives the materials which which cannot be recycled economically.

- In most European countries, there is now legislation which requires certain waste streams to be recycled.

Changes in Waste Stream Composition

With a significant change in the ocean incineration clientele, it is not surprising that there has been a corresponding change in the waste stream composition. Table 1

summarizes the change in chlorine and water content during the various periods of the industry.

Table 1

Chlorine and Water Content of Ocean Incineration Waste Streams

Period	% Chlorine	% Water
1972	40-70	<5
1976-79	30-40	<10
1980-83	30-35	10-30
1984-87	20-30	30-40

In the early years of the ocean incineration industry, most of the chlorinated waste originated from the production of chlorinated hydrocarbons and their derivatives, such as vinyl chloride monomer, ethylene dichloride and pesticides. In fact, in 1972 100% of the waste was generated from VCM/EDC production. After a few years some of the large generators installed on-site treatment facilities and thus caused a decrease in the waste streams with very high chlorine content.

In the early 1980's legislation and more rigid enforcement caused more wastes to be disposed of by incineration at sea, particularly wastes generated from the use of chlorinated solvents. These streams were characterized by a lower chlorine content resulting from a mixture of non-chlorinated

solvents, water and other impurities.

The average water content increased further in 1984/85 with a corresponding decrease in the chlorine content. This was effected by the promulgation and enforcement of regulations in northern European countries restricting the discharge of wastewater.

Technical advancements in the incineration process allowed for additional increases in the average water content during the past two years. Emphasis on recycling also had an impact on the waste composition. The average heat content of the waste decreased as more solvents found their way to recovery systems or supplemental fuel. However, solvents with relatively high water content could not be economically recycled and were therefore incinerated. For economic and technical reasons, the maximum water content for incineration on the Vulcanus is now limited to 40% -- considerably above the earlier limitation of 5%.

Changes in the Incineration Technology

Outside forces such as new regulations, stricter enforcement, active environmentalists and changing corporate policies have had, as explained above, an influence on the changing waste streams incinerated at sea. However, the largest impact has probably been due to technological advances in the incineration process.

The waste analysis procedure of the incoming waste has been improved considerably. The increased quantity and quality of waste stream data has allowed for better characterization. This then allows for better operational control during the loading and subsequent incineration process.

The waste feeding system on board the Vulcanus vessels has been modified to give more flexibility. In earlier operations waste from a single tank was feeding all three burners of a particular incinerator. The only exception was in the rare occasion when supplemental fuel was necessary. Then one or two burners could be switched to fuel. The new manifold system coupled with the better waste stream characterization now makes it possible to feed a single incinerator from as many as three different tanks.

Another major improvement to the overall incineration process has been a new high speed-computer. Now pre-alarms are actuated by much smaller changes in the monitoring system. For example, a system variance of 1 ppm in the carbon monoxide level will effect a change in the operation to correct even a small upset. The analog recorders of variables such as carbon monoxide, carbon dioxide, excess oxygen and temperature are now very smooth compared to earlier operations.

In addition to the computer, the operations have been improved by installing more sensitive flame arrestors. Operational changes are now controlled by small differences in the

flame color.

The Vulcanus II, in operation since 1983, was designed on the basis of a decade of experience with the Vulcanus. Since 1983, however, there have been so many improvements on both ships that they can now both be called a new generation.

Impact on the Market and Needs Determination

The changes in the ocean incineration industry described above -- generator base, waste stream composition and technological advances -- have certainly had an impact on the market in Europe. More different waste streams from more different generators are being incinerated on the Vulcanus vessels. This trend is expected to continue unless, of course, there are some regulatory or political roadblocks.

What impact, however, will these changes have on the outlook for ocean incineration in the U.S.? It certainly should be positive although the degree to which it will affect the U.S. program is unknown. It is impossible to assess the technical improvements necessary to compensate for adversity in terms of politics or perception. It is probably safe to state that the political environment or perception for certain issues is such that no degree of technological advances could outweigh negative attitudes. To present a meaningful argument here, one must assume that the regulatory process is administered and governed by prudent decision makers.

EPA conducted a very comprehensive market assessment as part of its March, 1985 report (EPA, 1985a). The study looked at liquid organic hazardous waste (LOHW) that would require incineration as a result of the land disposal bans, specifically looking at landfills, surface impoundments, boilers and deepwell injection. While each of these segments contributed to the total increase in the demand for incineration, the largest contributor was the deepwell injection segment. The LOHW stream from this segment would also be most affected by the technological advances in the ocean incineration process. Consequently, only the deepwell injection waste streams will be discussed in more detail to make the point.

EPA estimated (in 1985) that 50% of the waste being injected in deepwells would be affected by the land ban regulations.

Of this quantity, 5% could be incinerated directly while an additional 30% would require treatment and be reduced to a volume such that 5% of it would be subsequently incinerated. This resulted in 6.5% of the affected quantity, or 3.25% of the total quantity, going to incineration. EPA then applied these assumptions to the 1981 data of 32 million MT of waste injected in deepwells. The result was an additional one million MT requiring incineration.

Later in 1985, EPA published another report (EPA, 1985b) on injection of hazardous waste that was based on 1983 data.

Now the total volume being injected was 57 million MT. Applying the same assumptions as above results in 1.85 million MT requiring incineration. Regardless of which number is used, the potential increase in LOHW for incineration greatly exceeds the available capacity.

The underlying assumptions that were used to arrive at these estimates were based on the organic concentration levels of the various injected waste streams and the specification requirements of the Vulcanus vessels. At that time the ships could not accept any LOHW that contained more than 5% water. As mentioned earlier, the Vulcanus vessels can now incinerate waste streams containing as much as 40% water. Because it is very difficult to do so, no attempt is made here to estimate how much more LOHW would now be available for incineration at sea. The main point is that the available window has increased from 5% water to 40% water which certainly implies that the market for incinerable waste from injection wells is much larger.

Some commenters to the proposed ocean incineration regulations (of February, 1985) found EPA's conclusions of a shortfall in liquid landbased incinerator capacity to be incorrect. They cited a Congressional Budget Offfice (CBO, 1985) study on changes in waste management resulting from the 1984 Hazardous and Solid Waste Amendments (HSWA). They pointed out that this study showed the HSWA would cause the

volume of liquid hazardous waste to fall dramatically and therefore no shortfall would occur. More specifically, the quantity of waste to be disposed of in an injection well in 1990 will decrease 24% from from the 1983 quantities if no waste reduction is implemented. With waste reduction the decrease will be 38%.

Two important points must be made relative to this argument against a need for additional liquid incineration capacity. One, the CBO starting point for the estimates on future quantities of waste disposed by deepwell injection is 66.8 million MT. Even with a reduction of 38% by 1990 the quantity is 41.4 million MT which is much greater than EPA's starting volume of 32 million MT. Whatever number is correct (if any) -- 32, 57, 66.8 or 41.4 -- the potential increase in liquid incinerable waste is large. Second, the CBO study estimates that HSWA will decrease the volume of waste going to deepwells by 16 million MT <u>without</u> waste reduction. This quantity must be treated and/or disposed in some other manner, and a good likelihood is that a large percentage will be eventually incinerated.

CONCLUSION

The hazardous waste industry, still being in its infancy, is in a constant flux. Promulgation of new regulations continues. Enforcement of the regulations is improving. Environmentalists seem to be more active. The public is becoming more aware. Each these factors has contributed to the source and make-up of the liquid waste incinerated at sea. Originating with two companies generating waste containing 60-70% chlorine with virtually no water, the incineration business for the Vulcanus vessels has grown to over 2000 generators producing waste that averages 27% chlorine and almost 40% water.

More significant, however, has been numerous advancements in the incineration technology. This has allowed for better control of the incineration operation particularly in light of the greater number of waste streams. These technical advancements were both necessary and the cause for the changing waste stream mix.

All of this leads to the main point of this paper. For the majority of 16 years of Vulcanus operations, the specifications for incinerable waste streams were relatively narrow. In just the past 2-3 years, the range of acceptable waste streams has increased significantly. Thus the available market for incineration at sea is greater than ever before. Critics of the need for incineration at sea should take note.

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A NEW METHOD FOR COLLECTING, TRANSPORTING AND INCINERATING ABOARD SHIP -- TOXIC MATERIALS AND HAZARDOUS WASTES

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ABSTRACT

Unlike the design and operation of all existing incineration vessels, this SeaBurn system has five unique characteristics:

1) Standard ISO tank containers (20 ft) are used to receive the waste at the generator's site and retain it during both overland and sea transport until, at the approved ocean burn site, the waste is sequentially discharged from the containers and injected into the ship's incinerator for ultimate destruction.

2) No waste is carried or piped below the main deck of the vessel so that in the event of a collision or sinking, the containerized waste can be readily removed without substantial leakage into the environment.

3) The incinerator is oriented horizontally to minimize the period of air-borne plume exposure and to facilitate the taking of instrument readings and exhaust gas samplings.

4) Seawater scrubbers are used: a) to saturate the exhaust gases in order to promote rapid assimilation and dispersion into the sea, and b) to cool the acidic gases to prevent them from reaching air currents where they may be carried in undesirable directions.

5) Since shoreside storage tanks are not required to hold or blend the waste for this vessel, the SeaBurn ship can serve many different ports, thereby reducing the overland movement of the wastes.

BACKGROUND

Stolt-Nielsen is the world's largest marine carrier of bulk chemicals. It has a fleet of over 50 major chemical tankers in international trade carrying every type of liquid product, most of them dangerous. But Stolt is more than an ocean carrier. It has its own terminals, its own rail cars, trucks and barges and -- what I would like to discuss in more detail in a moment -- it has over 1900 intermodal tank containers, again, the largest fleet of these units in the world.

There are dimensions to the company that extend beyond the transport of chemicals. Stolt's interests have encompassed sea farming since 1971, with fish hatcheries located throughout the world. We have a fleet of specialized vessels used for the underwater inspection and maintenance of transoceanic cables, pipelines, drilling platforms, and equipped with deepsea ROV's (i.e. remote operated vehicles). Into each business we enter, we bring innovative technology, topnotch equipment and the highest levels of performance from our crews and employees.

Over the past decade Stolt-Nielsen has expended appreciable resources and assigned technical personnel to work with IMO and US Coast Guard to help draft measures to curtail marine pollution -- to eliminate or reduce the discharge of oil and oily wastes from ships at sea. These residues and tank washings must now be discharged ashore. Many of these waste

streams are candidates for incineration. At the same time, with the passage of US environmental statutes such as RCRA in 1976, generators of hazardous waste are now held responsible for disposing of these residues. In 1984, the landfilling of certain hazardous wastes became objectionable and was stopped by the state and federal regulatory authorities.

So, in response to the need for providing for disposal of wastes generated by both marine and shoreside activities, Stolt-Nielsen initiated the SeaBurn Project, the burning of toxic liquid wastes at sea.

SEABURN'S NEW APPROACH

Stolt's innovative approach towards ocean incineration was to introduce containerization in order to minimize handling the waste (ashore and afloat) and to introduce seawater scrubbers to control the drift of the exhaust plume at sea. In accomplishing these feats, we were also able to introduce several unique design features into the ship.

First we put all the containers on the main deck so we would not have to worry about crew safety in enclosed spaces throughout the ship. Also, this made loading and removing the containers from the vessel much easier by the shoreside crane operators. Everything was in full view.

It also gave us the freedom to locate the containers well inboard of the sides of the vessel so that, in the event of a

collision or sinking, the containerized waste could be readily retrieved by salvage crews without substantial release into the environment. The colliding vessel would have to penetrate deep into the side of the incinerator ship before any tank container could be struck. Even then, leakage would be limited only to those containers that were actually punctured.

SHIP DESIGN

Stolt has two configurations: an oceangoing barge having a capacity of 144 containers and 4 incinerators; and a self-propelled ship with 36 containers and 1 incinerator. In both cases the waste handling components of the system -- piping, containers, pumps, fans, incinerators, etc. -- are above the weather deck.

We laid the incinerators down horizontally and located them all the way aft so that the exhaust gases would be projected across the stern into the wake of the ship. This position then permitted the installation of a multi-phase seawater scrubber system having injector nozzles that shot the cold seawater deep into the superheated exhaust gas stream. Through this technique we are able to saturate the gases, thereby promoting rapid absorption into the sea and accelerating dispersion. It also further increases the already high density of the gas plume by cooling it, adding water vapor and removing its thermal lift. In this way, the plume will

remain low on the water until assimilation into the sea is quickly and fully completed.

The horizontal attitude of the incinerator also provides access along the full length of its barrel. The naval architect has thus provided a stable platform at the exit plane of the incinerator and elsewhere to take instrument readings on the performance of the continuing burn process.

THE CONTAINER

These containers have a capacity of 20,000 liters. In actual operations the allowable volume may be dictated by the density of the waste material and/or the axle load limitations of the highway regulations in the country being served.

The framework that surrounds the liquid-bearing part of the container not only acts as a barrier against physical damage to the tank, but it provides the structural means for lifting the container to and from the ship and securing it to the motor vehicle or railcar during the overland movement. This framework also provides the strength for stacking containers, one upon the other.

EFFECT ON THE SEA

Of any location where chlorinated hydrocarbons might best be destroyed, the ocean is the most "tolerant and forgiving". This is because the highly acidic effluents from the

incineration process are converted by the ocean action alone into salt, carbon dioxide and water vapor -- that is, the natural constituents of seawater -- and these are dispersd by the movement of ocean currents and kinetic energy (i.e., ship propulsion disturbances and wind and wave reactions). This is not the case with land based incinerators where caustic soda is needed to neutralize the gas emissions and, when the caustic is spent, it becomes itself a new toxic waste (and is frequently dumped through sewers into rivers flowing into the sea, thus increasing the amount of dissolved solids in the seawater, over and above what is produced by ocean incineration).

STATUS OF SEABURN

In October 1983 SeaBurn applied to the US EPA for an ocean incineration permit. The application is being held in abeyance pending the issuance of the EPA regulations, which are presently being finalized. Stolt will proceed with its plans as soon as practicable thereafter.

* * * * *

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THE ENVIRONMENTAL SAFETY AND ACCEPTABILITY OF INCINERATION AT SEA

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ABSTRACT

Numerous evaluations of incineration of organic materials at sea over the past 18 years have addressed issues ranging from the effectiveness and safety of this practice to opposition in principle based on the view that it is merely an inexpensive option which detracts from pollution control at the source.

The overall environmental safety and acceptability of incineration at sea was recently assessed by an international group of experts from the London Dumping Convention and the Oslo Convention (LDC/OSCOM/IAS 2/9, 27 April - 1 May 1987). The assessment showed no clear preference between incineration on land or at sea in terms of releases to the environment and the associated health or environmental risks. For example, there is no significant difference between organic emissions from at-sea or land-based incinerators when each is meeting the same destruction efficiency requirement. The assessment also showed the need for more information on the composition, persistence, toxicity and levels of organic emissions from all incinerators.

Environment Canada recognizes the deficiencies in knowledge of incinerator emissions and also the risk of spillage or accidental discharge. However, the understanding of the effectiveness and safety of incineration at sea is well known enough to accept it as a viable technology for handling liquid hazardous wastes. Environment Canada is therefore prepared to consider issuing an Ocean Dumping Control Act permit for incineration at sea if an applicant can meet the regulatory requirements and if there are no practical alternatives that are environmentally preferable.

BACKGROUND

Canada has not incinerated chemical wastes using ocean incineration vessels and has no immediate plans to do so. We have received a preliminary application from Chemical Waste Management, Inc., owners of the commercially available ships Vulcanus I and II. We have also met with representatives from Seaburn Inc. to review their container-based incineration system which is considered to be a promising technology. Our priority at this stage is to prepare an operating procedures manual for incineration at sea destruction facilities.

We are also aware that despite numerous evaluations of incineration at sea over the past 18 years of operating experience, conflicting opinions still exist. Issues range from questions relating to the effectiveness and safety of marine incineration to opposition in principle based on the view that it is merely an inexpensive option which detracts from pollution control at the source.

ENVIRONMENTAL SAFETY AND ACCEPTABILITY

The overall environmental safety and acceptability of incineration at sea was recently assessed by an international group of experts from the London Dumping Convention and the Oslo Convention (LDC/OSCOM/IAS 2/9, 27 April - 1 May 1987).

Key concerns expressed about incineration at sea include the effects of organic emissions, the risk of accidental spills and the waste management role for incineration at sea.

Using the London Dumping Convention (LDC) regulatory level of 99.99 per cent destruction efficiency, and assuming an upper limit of 1 nanogram per cubic metre (ng/m^3) for emissions of dioxins and furans, and under a worst-case scenario in which the emissions enter the sea in a restricted area (for example, 150 metres x 1500 metres), the LDC-Oslo Convention experts calculated that concentrations in the upper 25 centimetres of seawater may reach 0.5 parts per trillion (ppt).

The worst-case scenario was considered pessimistic as most of the organic emissions would likely be present as an aerosol and would, therefore, be more widely dispersed. The maximum concentration of dioxin and furan emissions measured during a PCB test burn at sea was 0.1 ng/m^3. Dioxins and furans are composed of compounds of widely varying toxicities and the most toxic compound, 2 3 7 8 tetrachlor-dibenzo-p-dioxin (TCDD), is only a small component. Finally, records of incineration at sea operations show a normal operating destruction efficiency of 99.9999 per cent.

Questions do, however, remain and more information on the composition, persistence, toxicity and levels of organic emissions is required and will be pursued for both land- and sea-based incinerators.

The expert group examined the risks of transport of hazardous wastes and recognized that the likelihood of spills during incineration at sea operation would depend on a number of factors such vessel design, loading facilities, weather conditions, location of the incineration site, etc. Although no major spillages from chemical tankers during open sea transport of hazardous substances has occurred, it is generally accepted that such risks do exist and could result in major effects on the marine environment. The expert group concluded that it is important for the effects of spills be reviewed in the context of their probability and a separate expert group will be established to examine the risks of spillage from incineration at sea.

In assessing incineration at sea, the expert group also examined the practical availability of alternative land-based methods of treatment, disposal or elimination for the wastes concerned. In this connection, the LDC requires that alternatives to incineration of wastes at sea must be considered in the light of a comparative assessment of:

- Human health risks;
- Environmental costs;
- Hazards (including accidents) associated with treatment, packaging, transport and disposal;
- Economics (including energy costs);
- Exclusion of future uses of disposal areas

for both incineration at sea and the alternatives.

Overall, the LDC-Oslo assessment showed no clear preference between incineration on land or at sea in terms of releases to the environment and the associated health risks. For example, there is no significant difference between organic emissions from at sea or land-based incinerators when each is meeting the same destruction efficiency requirement. Because land-based incineration occurs relatively close to human populations, its primary risk is the potential for adverse impact on human health - resulting from exposure to routine or normal releases of waste or waste by-products. In contrast, ocean incineration's primary risk is to the marine environment because of the potential for a major accidental spill.

OCEAN INCINERATION PERMITTING PROCEDURES

Environment Canada has examined the available information and has found no data to indicate that incineration at sea is an unfavourable method of waste disposal. At the same time we recognize that public support is essential if incineration at sea is to go forward.

In evaluating specific proposals for incineration at sea, Environment Canada will be following the application review procedures outlined in Figure 1 (Ocean Incineration Permitting Procedures).

For an application to be considered complete, descriptions must be provided for the following elements:

- waste transportation, handling and storage;
- equipment operation and maintenance;
- waste analysis and test methods;
- monitoring and record keeping;
- cleaning operations;
- contingency plans;
- liability insurance; and
- reason for incineration at sea compared to other options.

Completed applications will be distributed to other involved and interested parties, such as the Canadian coast and provincial environmental agencies, for their comments.

The ODCA permit requirements parallel the LDC regulations and include provisions for site specific requirements. The LDC regulations, which Canada helped to draft, are considered to provide an effective level of control and protection. Specific standards exist for:

- vessel certification (e.g., IMO Type II Bulk Chemical Carrier);
- wastes requiring special studies (e.g., PCBs);
- performance (Destruction and Combustion Efficiency);
- recording devices and records;
- transportation and storage;
- contingency plans; and
- cleaning operations.

Ocean incineration would be deemed acceptable and a draft permit prepared if the ODCA regulatory requirements could be met and if it posed less or no larger human health and environmental risks than practicable land-based alternatives.

Canada would require an ODCA loading permit and a LDC dumping permit, or similar approval, before allowing export of hazardous wastes for incineration at sea. In addition, if the dump site were under the national jurisdiction of a third party country that objected to the planned dumping operation Canada would not issue an ODCA loading permit.

No permit for incineration at sea would be granted if the operation were prohibited under any other act of parliament or if a licence or permit required under any other such Act has not been obtained. This contributes to a comprehensive waste management approach. Applicants are specifically informed that they require provincial approval for the land-based collection, transportation and storage of waste. Transportation of wastes also involves the federal Transportation of Dangerous Goods Act.

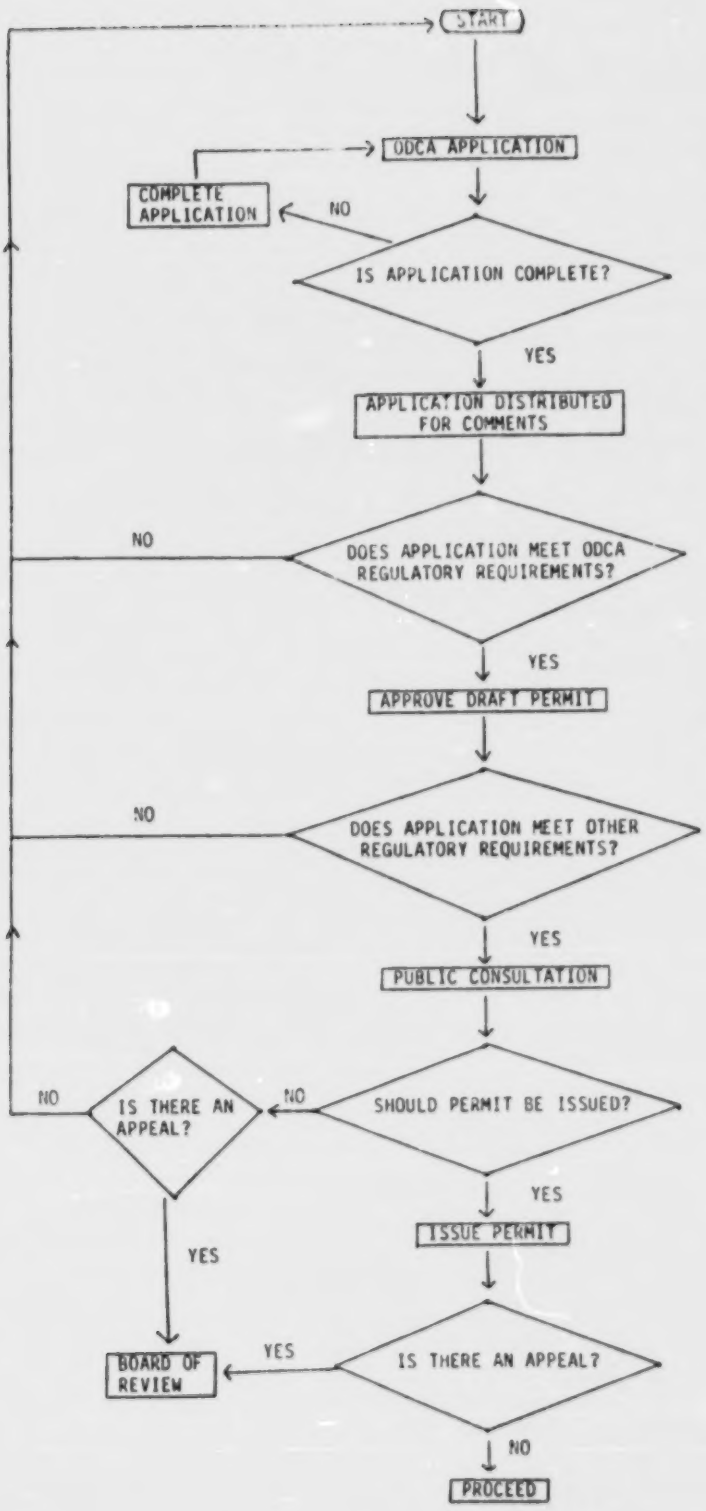

FIGURE 1 OCEAN INCINERATION PERMITTING PROCEDURES

All ocean dumping permits and amendments to permits must be published in the Canada Gazette before they can be implemented. If a permit is refused, or if the applicant is dissatisfied with the operating conditions specified therein, he may appeal and call for a Board of Review. A Board may also be established to hear objections from the general public to the granting of any permit, or the terms under which it is issued.

Inspections are conducted to verify compliance with the terms of a permit. Inspectors appointed under the ODCA may seize or detain any ship suspected of violating any provision of the Act or regulations, including the terms and conditions of a permit. If the owner is convicted, the vessel may be forfeited. Besides having to pay a fine of up to $100,000 for each offence, he may be billed the cost of any clean-up operation or other remedial action taken by the Crown. In addition, any applicant may be required to provide a monetary assurance (e.g., posting of bond or other asset) to the regulatory agency.

CONCLUSION

Environment Canada recognizes the deficiencies in knowledge of incinerator emissions and also the risk of spillage or accidental discharge. However, the understanding of the effectiveness and safety of incineration at sea is well known enough to accept it as a viable technology for handling liquid hazardous wastes. Environment Canada is therefore prepared to consider issuing an Ocean Dumping Control Act permit for incineration at sea if an applicant can meet the regulatory requirements and if there are no practical alternatives that are environmentally preferable.

Environment Canada also believes that the need for waste treatment and disposal options such as incineration at sea will continue due to the existing and future quantities of wastes, the time required to implement waste reduction measures and to develop sufficient capacity for recycling and advanced treatment, and the fact that not all wastes will lend themselves to such technologies. Incineration at sea is not viewed as detracting from the establishment of land-based incineration alternatives or the development of waste recovery and recycling technologies.

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ECOLOGICAL AND HUMAN HEALTH RISKS FOR SEWAGE SLUDGE DISPOSAL AT THE 106-MILE SITE.

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ABSTRACT

Methods for conducting ecological and human health risk assessments for sewage sludge disposal at the 106-Mile Ocean Disposal Site are based on estimates of mass loading rates to the site, and couple exposure and effects assessments on compatible time scales. Short-term (time-variable) exposure calculations assess potential acute effects. Long-term (time-averaged) calculations are used in assessments of potential chronic effects. Exposure calculations can be based on any of four models: (1) a short-term time-variable model for dispersion in the first few hours after discharge, (2) a long-term time-averaged model for obtaining upper-bound estimates of exposure in the upper mixed layer, (3) a long-term accumulative model for obtaining upper-bound estimates of contaminant build-up in sediments, and (4) a three-dimensional Lagrangian-Eulerian model for obtaining realistic estimates for water column and sediment exposure.

Calculations for ecological risk are based on simple ratios of predicted water column concentrations to EPA water quality criteria. Although estimates of sediment concentrations can be calculated, sediment quality criteria currently are not available. Uncertainties make it difficult to predict long-term, regional scale ecological effects in both the water column and benthos. For example, the relative sensitivity of nearshore vs. offshore species is not well known.

Calculations of human health risk are based on potential accumulation of sludge constituents in biota, which subsequently can be consumed by man. Tissue body burdens for pelagic species are computed using a bioaccumulation factor (BCF). For benthic organisms, tissue concentrations are calculated with a preference factor analysis which uses organic carbon content of sediments and lipid content of the organisms. Reference doses for

compounds judged to be noncarcinogenic are obtained from EPA's Integrated Risk Information System, and are used in conjunction with fisheries' landings and consumption data to evaluate noncarcinogenic risk to humans. Carcinogenic risk from fish consumption is based on estimated potency factors.

INTRODUCTION

The impetus for the recent deep-ocean modeling work at ERL-Narragansett (ERLN) was the decision in City of New York vs. EPA (BNA, 1981). This lawsuit resulted from the Agency's interpretation of the 1977 amendment to the Marine Protection, Research, and Sanctuaries Act which prohibited dumping of sewage sludge in ocean waters after 31 December 1981. In this decision, Judge Sofaer determined that the Agency did not have justification for denying sewage sludge dumping permits to New York City for the 12-Mile Site. Although sludge dumping continued at the 12-Mile Site, it was understood that the Agency intended to phase out use of this site in favor of the 106-Mile Ocean Disposal Site (Figure 1). ERLN recognized in 1983 that if sludge dumping were to be moved to the 106-Mile Site, assessment methods would be needed to evaluate permit applications and to determine impacts associated with this disposal of wastes. Because of the different oceanographic environment that would be encountered off the continental shelf, the assessment methods used for nearshore disposal sites would not be appropriate for deep-ocean sites.

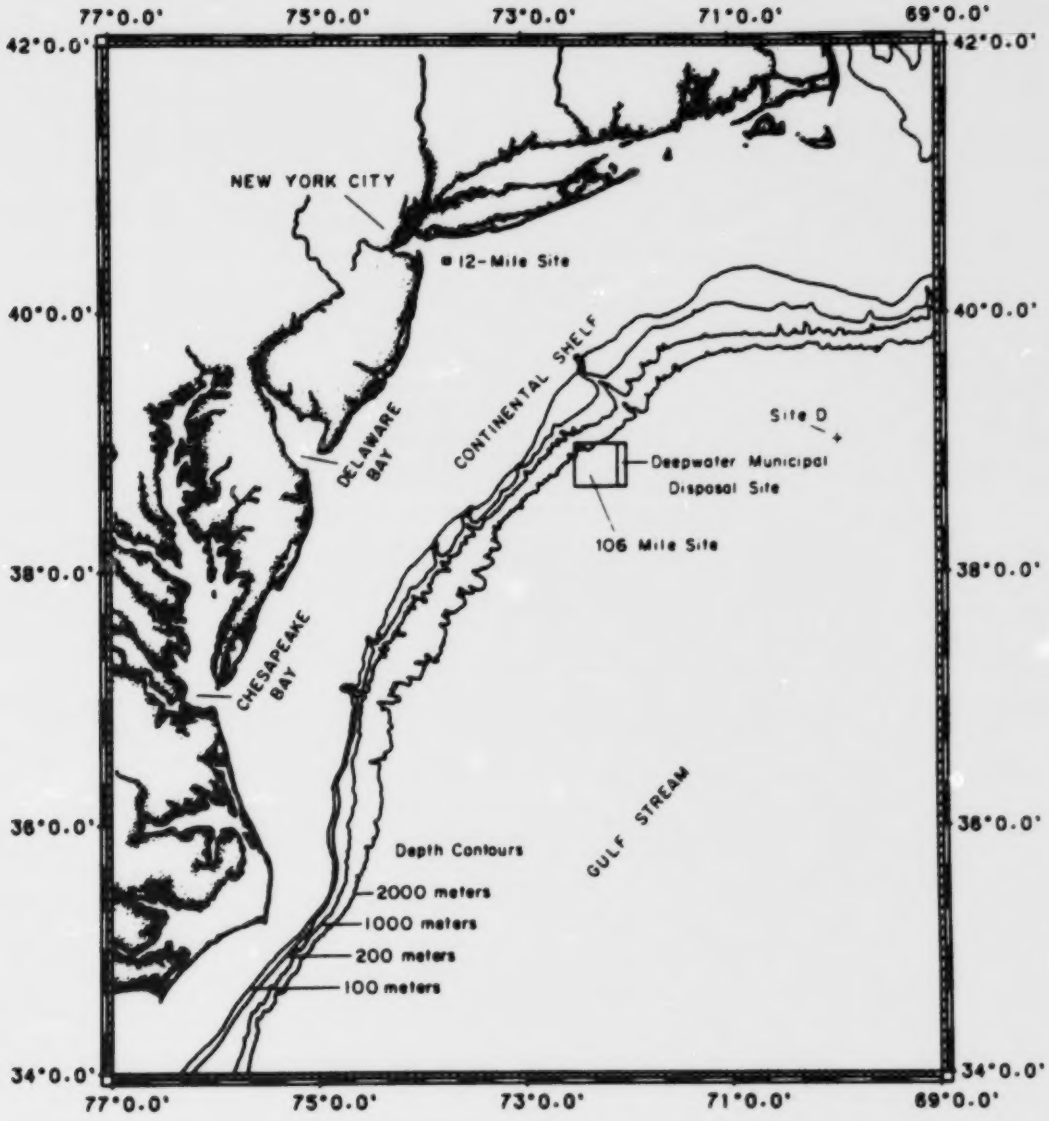

Figure 1. Location of the 106-Mile Ocean Disposal Site.

ERL-NARRAGANSETT RISK ASSESSMENT FRAMEWORK

The risk assessment framework used at ERLN for addressing potential impacts of municipal sludges disposed at the 106-Mile Site is shown in Figure 2. This framework is a flow diagram starting with the introduction of wastes into the ocean, following various pathways to the ecological and human health endpoints.

Exposure calculations have been based on one of four models: (1) a short-term time-variable model for dispersion in the first few hours after discharge, (2) a long-term time-averaged model for obtaining upper-bound estimates of exposure in the upper mixed layer, (3) a long-term accumulative model for obtaining upper-bound estimates of contaminant build-up in sediments, and (4) a three-dimensional Lagrangian-Eulerian model for realistically estimating water column and sediment exposures.

Parameters used to address ecological risks are acute or chronic toxicity values and water quality criteria. Information from bioassays conducted with whole waste provides estimates for LC50s or EC50s. Diluted concentrations of individual contaminants in the waste are compared with existing EPA marine water quality criteria on compatible temporal scales. Ecological risk assessment as conducted here is not a true quantitative assessment; rather, we are conducting a qualitative assessment by

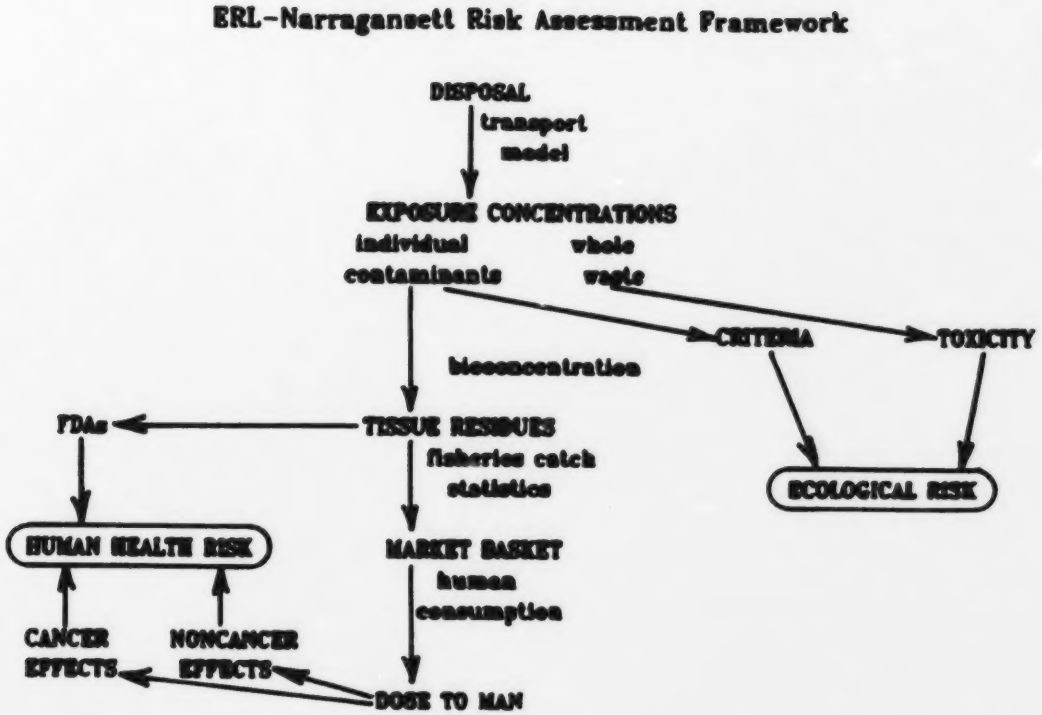

Figure 2. Risk assessment framework for ocean disposal.

comparing predicted exposure levels with available criteria in a pass/fail test. Bioaccumulation of individual contaminants in marine species (tissue residue levels) is a human health concern (i.e., ingestion of contaminated organisms). Insufficient information presently is available to link tissue residue concentrations with specific biological effects in exposed organisms.

Human health criteria include the U.S. Food and Drug Administration (FDA) tolerance and action levels for consumption of fish and shellfish, and estimates of carcinogenic and noncarcinogenic effects. Comparisons with FDA levels and noncarcinogenic effects provide a qualitative pass/fail test, while estimates of carcinogenic effects provides a quantitative estimate of human health risk.

Another component of EPA's research program associated with ocean disposal of sewage sludge at the 106-Mile Site involves design and implementation of a monitoring program. A basic assumption of the ocean dumping permit program is that compliance with permit conditions is sufficient to protect the marine environment. In practice, permitting decisions for individual applicants are often based on considerations of short-term, nearfield compliance with both toxicity and water quality criteria. Issues related to potential long-term ecological and human health impacts related to the total mass loading from all disposal activities are more difficult to address. The monitoring program is being designed to test compliance with permit

conditions, and also to determine if dumping under the permit conditions is sufficient to provide an adequate margin of safety for the marine environment.

RISK ASSESSMENT: UPPER WATER COLUMN APPLICATION

The original application of the ERLN risk assessment of waste disposal at the 106-Mile Site was for long-term, time-averaged upper water column impacts (Paul et al. 1987). This addressed risk on the temporal scale of months and the spatial scale of 10-300 km. Subsequently, this application was expanded to include a short-term (hours to days) risk assessment on the scale of the disposal site. The major assumption in these applications is that contaminants in the waste are completely dissolved and remain uniformly mixed in the upper mixed layer of the ocean; this provides an upper-bound estimate of risk for these applications.

It is important in any risk assessment that exposure and effects (hazard) determinations be conducted on compatible scales. In our applications, short-term acute effects endpoints are considered in conjunction with expected short-term exposures, whereas chronic effects and bioconcentration endpoints are considered in conjunction with long-term exposures. In a complete risk assessment, optimal disposal operations for environmentally safe management of the site must assure that both short-term and long-term exposures are within acceptable limits for the criteria used.

Current regulatory practices for ocean disposal of municipal sludges at the 106-Mile Site address short-term toxicity and two types of water column concentration based criteria: (1) criterion maximum concentration (CMC) designed to protect against adverse acute effects, and (2) criterion continuous concentration (CCC) designed to protect against adverse chronic effects. Toxicity and acute criteria are met by regulating discharge rates (e.g. sludge volume per minute discharged into vessel wakes) as stipulated in disposal permit conditions. Chronic criteria are met if the site is sufficiently dispersive to accommodate the contaminant specific loading rate (e.g. mass per day) to the site. Short-term and long-term applications of an upper water column risk assessment for sludge disposal at the 106-Mile Site are given in Walker et al. (1987). Discussion here will focus on an assessment of risks related to projected contaminant mass loading rates to a site and potential chronic exposures in the upper water column.

Long-term exposure calculations are based on an annual loading of 7.2 million wet metric tons of sludge to the 106-Mile Site, which was the 1980 annual disposal at the 12-Mile Site. Sewage sludge concentrations for Cu and PCB are 54 and 0.05 μg/g, respectively. Equivalent loading rates are 1000 and 0.98 kg/day respectively (Walker et al., 1987). These compounds are chosen for illustration because Cu is a toxic sludge constituent and PCB is a suspected

carcinogen and accumulates in the tissue of some valuable aquatic organisms.

Figure 3 illustrates the area over which long-term elevations above background concentrations are predicted. The same spatial region displays sludge dilution (Figure 4). Long-term dilutions range from 1×10^6 to 1×10^7. Whole-sludge toxicity ratios are calculated by dividing predicted long-term average dilutions by the dilution factor of 1×10^5, which is required to achieve the limiting permissible concentration (LPC). This LPC is based on an LC50 concentration of 0.1 percent for the copepod Eurytemora herdmani with sludge collected from the Middlesex County Utilities Authority, Sayerville, New Jersey (Miller et al. 1987). If all of the sludges from metropolitan New York and New Jersey were this toxic, predicted whole-sludge toxicity safety ratios would range from 10 on the scale of the 106-Mile Site to 100 over a broad area downstream of the disposal site (Figure 4). Passaic Valley Sewage Authority is the largest New Jersey discharger and is 60% industrial by volume. Sayerville is predominantly residential and may not represent the worst case. Many of the sludges from New York City municipal treatment plants have LC50 values that correspond to approximately 1% whole sludge in bioassays using the mysid Mysidopsis bahia (New York City Department of Environmental Protection, 1983).

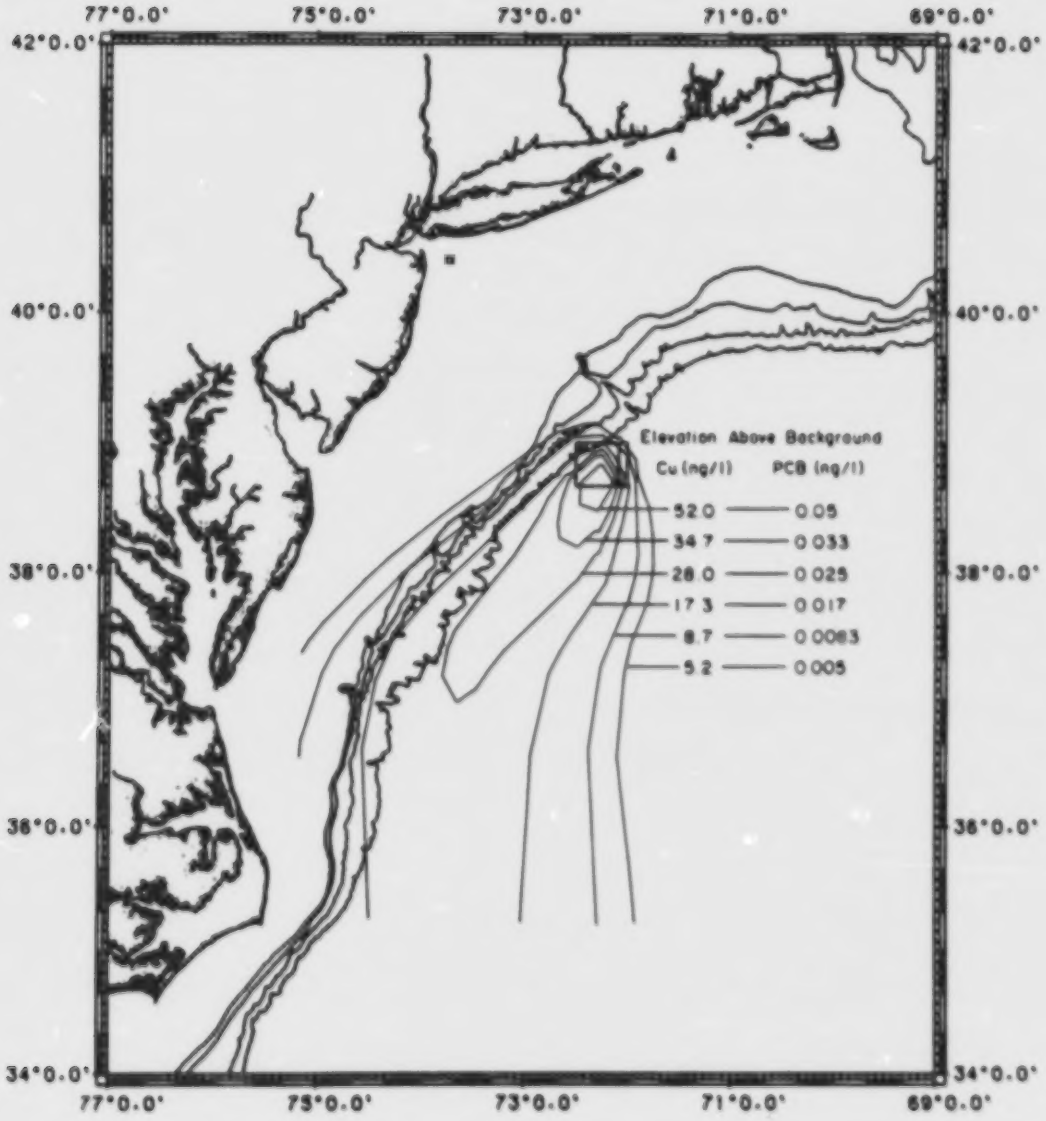

Figure 3. Areas in upper mixed layer in which long-term elevations in sludge constituents are predicted.

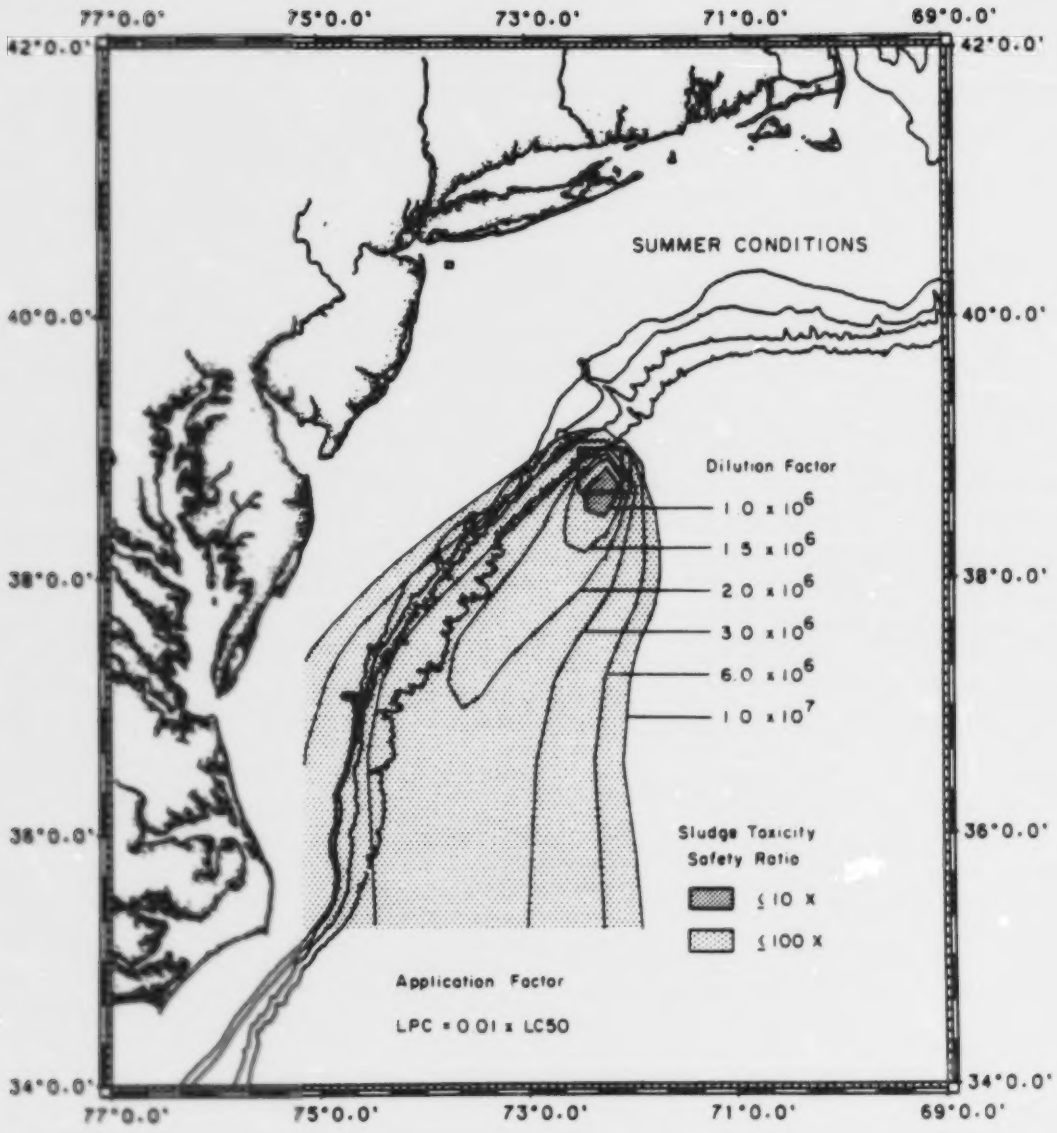

Figure 4. Long-term time-averaged dilutions in the upper mixed layer and whole sludge toxicity safety ratios.

At the 106-Mile Site, Cu background concentration is approximately 170 ng/L, and New York/New Jersey sludges contain approximately 54 μg/g total copper on a wet weight basis. If sludge is initially diluted by a factor of 100,000 (to meet the LPC for a 0.1 percent LC50), the Cu concentration will be elevated above background by 0.54 μg/L, for a predicted total Cu concentration of 0.71 μg/L. Because the calculated total Cu concentration for sludge diluted by a factor of 100,000 is well below both the EPA marine CMC for total Cu (23 μg/L) and the EPA marine CCC for total Cu (4 μg/L), no acute or chronic copper toxicity is predicted.

PCB background concentration of the 106-Mile Site is estimated to be approximately 0.05 ng/L (O'Connor et al., 1985). New York/New Jersey sludges contain approximately 0.05 μg/g (wet) total PCBs. For dilution by a factor of 100,000, PCB concentrations are elevated 0.5 ng/L, for a calculated total PCB concentration of 0.55 ng/L. Available data indicate that acute toxicity to saltwater aquatic life probably occurs only at concentrations above 10 μg/L. The predicted PCB concentration is also below the marine CCC (0.030 μg/L). Therefore, no acute or chronic PCB toxicity to marine organisms is expected under these disposal conditions.

Long-term assessment results for predicted PCB tissue residues are illustrated in Figure 5. Safety ratios are calculated by dividing the FDA tolerance level of 2 ppm by the predicted long-term, time-averaged tissue residues (based on long-term, time-averaged water column total PCB concentration and a bioconcentration factor of 5×10^5). Safety ratios for the upper mixed layer range from a factor of 40 on the scale of the disposal site to a factor of 70 over a broad section of the slope water region based on this criterion.

RISK ASSESSMENT: BENTHIC APPLICATION

Since contaminants accumulate in marine sediments, the time scale appropriate for the benthic risk assessment is much greater (10-100 years) than those used in upper water column risk assessment (seasons). A benthic application of the ERLN risk assessment for noncarcinogenic human health effects related to waste disposal on and off the Middle Atlantic Bight continental shelf is presented in Nocito et al. (1987). These calculations have been extended to include an assessment of carcinogenic risks in a companion paper in this symposium proceedings (Nocito-Gobel et al., 1987). The contaminant transport model for calculating flux through water and into the sediments assumes that all contaminants are associated with particulates. Since degradation losses and resuspension are not explicitly included in the model, results represent upper-bound

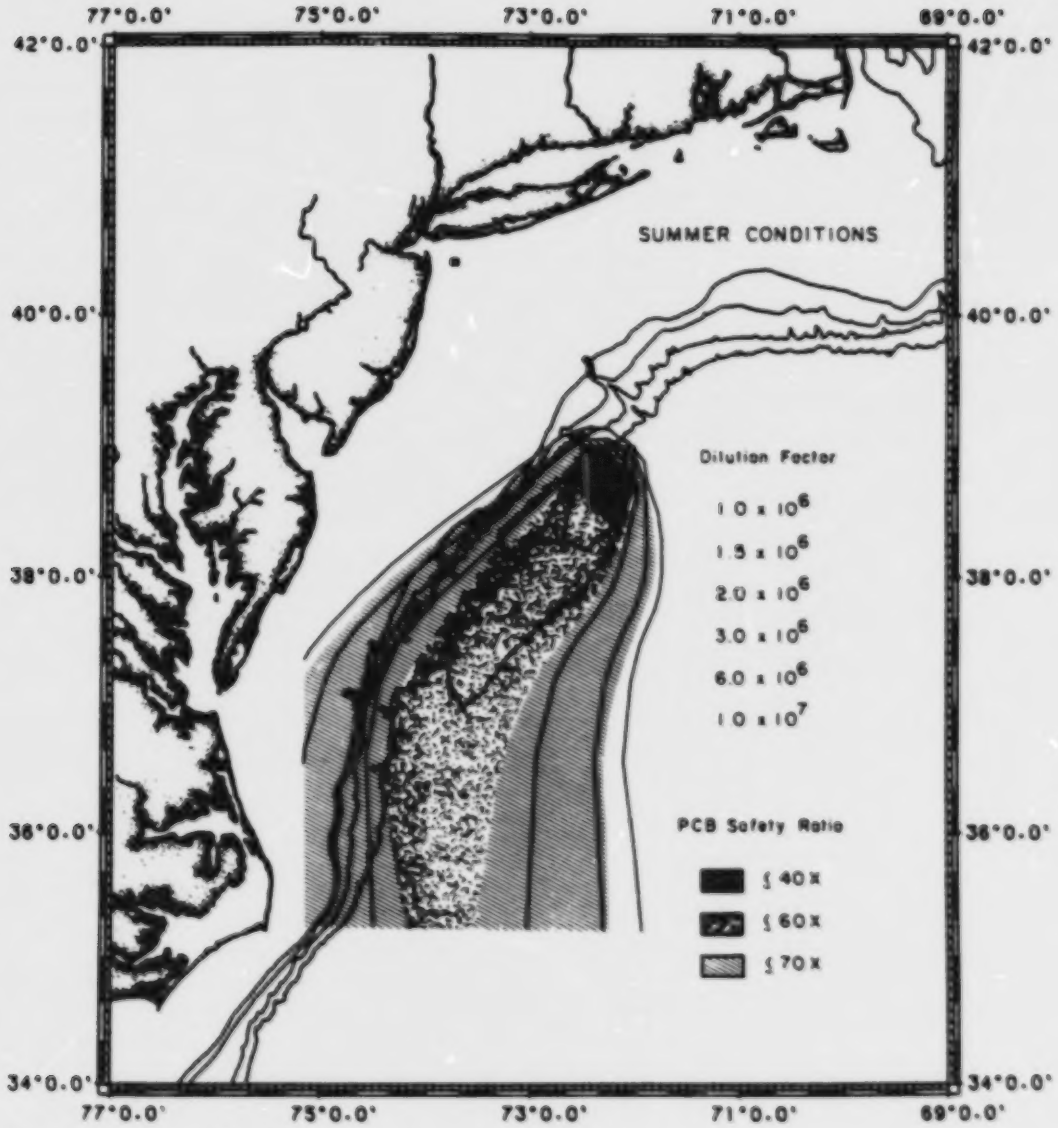

Figure 5. PCB safety ratios, based on the FDA tolerance level of 2 ppm.

estimates of contaminant flux to the sediments. Residues in benthic species were used as contaminant vectors to demersal fisheries and then to man to assess human health risk (Nocito-Gobel et al., 1987). Ecological risk assessments for the benthos were not conducted because there are no marine sediment quality criteria.

RISK ASSESSMENT: THREE DIMENSIONAL APPLICATION

The work described above provided upper-bound estimates of exposures for the upper water column and sediments. These estimates assume the contaminants are either completely dissolved or particulate-bound. For realistic estimates of exposures for water column and sediments, one must consider contaminant partitioning between dissolved and particulate phases. This work is in progress. A brief description of the approach follows.

A Lagrangian-Eulerian approach determines the three-dimensional distribution of contaminants in dissolved and particulate phases through the water column and into sediments (Paul et al., 1986). The Lagrangian calculations determine three-dimensional waste particle trajectories. Horizontal motion is determined from long-term current meter data records. Verticle particle motion is due to mixing (turbulence) and gravitational settling. An ensemble of particle trajectories is used to calculate time-averaged distributions of the material in a Lagrangian frame of reference. The spatial distribution of particles is then

transformed to an Eulerian frame. Horizontal dispersion is then calculated by an inverse procedure using an Eulerian convective-diffusive equation, and the particulate and dissolved phase contaminant distributions are solved using equilibrium partitioning theory. Once the three-dimensional distributions in the water column and sediments are calculated, both human health and ecological risk assessments can be determined.

SUMMARY

Current regulatory practices for municipal sludge disposal at the 106-Mile Site employ short-term toxicity and pollutant concentration based water quality criteria. Risks associated with short-term exposure can be managed by regulating discharge rates. To determine if disposal practices are apt to lead to violations of marine criterion continuous concentrations, the ERLN approach for assessing risks incorporates a consideration of dispersive characteristics of the site and estimates of contaminant specific mass loading rates to the site. Based on assessments for 15 pollutants of concern to the EPA Office of Water Regulations and Standards, no ecological water quality criteria violations are predicted for the upper mixed layer of the water column based on projected loading rates (Walker et al., 1985).

Considerations of potential human health impacts have involved comparison of projected tissue residues with FDA action or tolerance levels, but also are based on dose to man calculations which utilize estimates of human consumption rates of contaminated aquatic organisms. It is possible to calculate upper-bound estimates of potential human health risks due to the presence of specific persistent contaminants. This is done in a companion paper (Nocito-Gobel et al., 1987) in this symposium proceedings volume.

Models developed at ERLN estimate upper-bounds of contaminant exposure in either the upper water column or sediments. However, these methods should only be used as screening tools to raise cautionary flags concerning potential risks. If these flags are raised it is important to refine exposure calculations. To do this with sufficient precision and accuracy, we need to understand processes controlling vertical transport, degradation losses, sediment transport, and bioavailability of contaminants. This type of information can be factored into the three-dimensional Lagrangian-Eulerian exposure calculations that are being developed.

ACKNOWLEDGMENT

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Approval does not signify that the contents necessarily reflect the views and policies of the U.S. Environmental Protection Agency. This is Contribution No. 887 of the Environmental Research Laboratory - Narragansett.

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COMPARATIVE RISK ASSESSMENT FOR MARINE DISPOSAL OF SEWAGE SLUDGE AT MIDSHELF AND OFFSHELF SITES

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A risk assessment framework is presented for predicting impacts from midshelf and offshelf disposal of sewage sludge. In the application presented, only impacts resulting from bioaccumulation of contaminants in tissues of organisms exposed via the benthic food chain are considered. Disposal sites are assumed to be characteristic of sites in the Middle Atlantic Bight. The contaminant transport model used assumes that all contaminants are associated with particulates, and calculates flux through the water column and into bottom sediments. Since degradation losses and resuspension are not included in the model explicitly, results obtained represent upper-bound estimates of time-averaged contaminant flux to the sediments. Cross-shelf profiles of maximum concentrations in the sediment surface layer are calculated using an expression which accounts for mixing in the bioturbated zone. Assuming that organisms reach equilibrium with surrounding sediment, concentrations of neutral hydrophobic organic contaminants in infaunal organisms are computed using a preference factor analysis, which is based upon organic carbon content of the sediments and lipid content of the organisms. Results indicate that body burdens may be at least half an order of magnitude higher in organisms living near the midshelf site than those living offshelf. Predicted tissue residues are compared to the FDA tolerance level for PCBs and the action level for chlordane. Using reference dose value obtained from EPA's Integrated Risk Information System for chlordane, non-carcinogenic risks to humans are computed based on a dose to man calculation. Carcinogenic effects are determined using estimated potency factors. Based on relative comparisons in this analysis, higher human health risks are associated with midshelf disposal.

Introduction

In 1984, the U.S. EPA designated a portion of the 106-Mile Site for sewage sludge disposal (EPA, 1984a). This is the only deep-ocean site in the U.S. designated for this purpose. As more restrictions are placed on land-based waste disposal sites, municipalities other than New York are beginning to consider application for ocean disposal permits (T. O'Connor, personal communication). The suitability of using sites on the continental shelf rather than deep-ocean sites is being considered. To address this issue, EPA needs to develop a method for evaluating the relative impacts of midshelf vs. offshelf disposal.

A comparative analysis of potential impacts resulting from midshelf and offshelf marine disposal of sewage sludge is presented in a risk assessment framework. In this application of the risk assessment framework, residues in benthic species are used as contaminant vectors to demersal fisheries and then to man to assess human health risks associated with sewage sludge disposal. The primary objectives of this paper are: (1) to present a framework which would have utility for performing assessments of potential risks from disposal of sewage sludge due to benthic accumulation of persistent contaminants, and (2) to compare the relative impacts of disposing sewage sludge at midshelf vs. offshelf sites. After presenting a brief description of the risk

assessment framework, the risks associated with midshelf vs. offshelf disposal of sewage sludge containing two persistent hydrophobic organic compounds, PCB and chlordane, are compared. Due to the potential for long-term benthic accumulation of contaminants as a result of ocean disposal, the time frame used in this assessment is 10 to 100 years.

Risk Assessment Framework

The risk assessment framework used for ocean disposal is based on the work of Paul et al. (1987). To evaluate the hazard associated with exposure to toxic compounds, risk assessment involves the following: two information gathering steps (site and waste characterization), followed by three evaluation steps (exposure assessment, hazard assessment and risk characterization). Information concerning site and waste characteristics is used in all three evaluation steps. Exposure assessment relates the transport and fate of source input of contaminants to resultant concentrations in the environment. Hazard assessment is used to relate environmental concentrations to biological effects, such as toxicity to organisms and bioaccumulation. The risk characterization step combines the exposure and effects results to provide a scientific basis for estimating risks. When this procedure is applied to multiple sites, relative comparisons can then be made pertaining to potential impacts from waste disposal. The risk assessment framework developed at the

U.S. EPA Environmental Research Laboratory in Narragansett (ERLN) is illustrated in Figure 1.

A layered two-dimensional transport model was used to relate waste input to benthic flux of PCBs and chlordane into the bottom sediment. Estimated exposure concentrations in sediment were used to predict bioaccumulation in tissues of benthic organisms. Human health risks were evaluated based on U.S. Food and Drug Administration (FDA) tolerance and action levels for contaminants in tissues, and doses to man to assess both carcinogenic and noncarcinogenic effects (Figure 1).

Site Characteristics

Synoptic current meter data from the Mid-Atlantic Slope and Rise (MASAR) Physical Oceanography Study (SAIC,

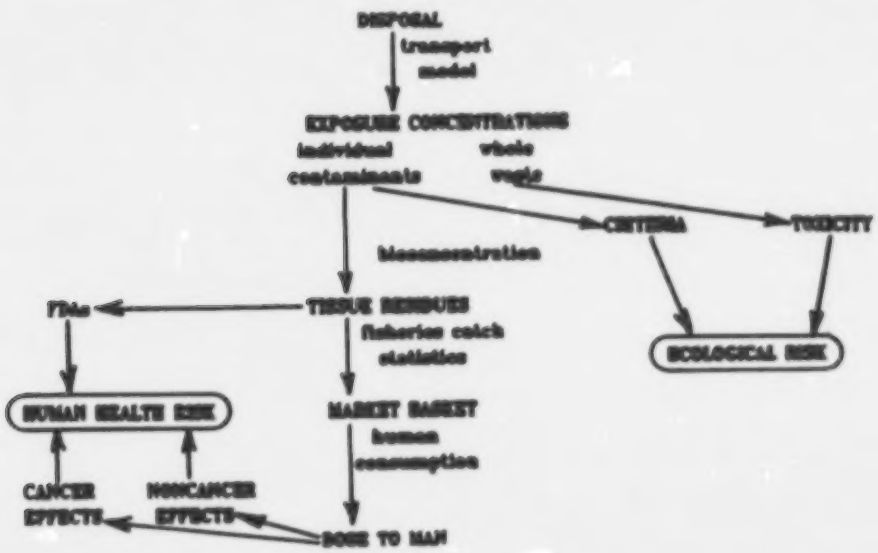

Figure 1. ERL-Narragansett Risk Assessment Framework

1986) were used to simulate dispersal of waste plumes for two hypothetical sites. The midshelf simulation is based on data from approximately 70 km offshore along the 60 m isobath, and the offshelf simulation, 150 km offshore along the 2000 m isobath. The simulations are intended to characterize dispersion from sites in the Middle Atlantic Bight region. In this comparative assessment, we focus on cross-shelf variations to predict benthic fluxes. A detailed description of the site and waste characterization for this assessment is provided in Nocito et al. (1987).

Exposure Assessment

Description of Layered Two-Dimensional Model

The layered two-dimensional model is an extension of the modeling approach used in the long-term ocean disposal model developed at ERLN (Paul et al., 1987; Walker et al., 1987). This model provides upper-bound estimates of the contaminant flux through the water column and into sediments based on mass loading rate to the system. We used an annual mass loading rate of 7.3 million wet metric tons of sludge which reflects recent loading rate to the 12-Mile Site (Santoro et al., 1986). In the model, waste plumes are described as discrete two-dimensional Gaussian plumes that are advected by the mean current, dispersed according to the variance-covariance structure of the currents and settle according to a specified settling velocity distribution. We used data from settling column

experiments with sludges to determine a settling velocity distribution (Faisst, 1981).

To be consistent with the upper-bound calculation, the following model assumptions are made. All contaminants are associated with the particulate phase and are not degraded nor transformed. Resuspension is not simulated in the model. Topographic changes do not occur in the along-shelf direction. The midshelf site is modeled as a single layer and the offshelf as 3 layers consistent with the availability of MASAR current meter data.

Predicted Exposure Field

Long-term, benthic accumulation results were based on a particulate loading of 10^6 kg of solids/day; i.e. a volume of 20,000 cubic meters of sludge with 5% solids. The predicted maximum flux of 100 g solids/m^2/yr by disposal at the midshelf site was approximately one and a half orders of magnitude larger than that at the offshelf site (5.0 g solids/m^2/yr). Contaminant flux distributions were obtained by rescaling results using contaminant specific loading rates; namely, 2.57 kg/day of PCBs and 5.26 kg/day of chlordane (EPA, 1984b). Since the focus was on cross-shelf variations in this assessment, cross-shelf profiles of average contaminant flux to sediment then were calculated by averaging flux values in the along-shelf direction over a 500 km distance.

To determine sediment concentrations on the time scale considered in this assessment, the effects of benthic biological activity on sediment mixing needed to be accounted for. We used a bioturbation coefficient of 1.0 cm^2/yr for the midshelf site and 0.1 cm^2/yr for the offshelf site, both of which are at the low end of reported values (Nocito et al., 1987). Contaminant concentrations were averaged over the top 2 cm of sediment for determining exposures to the benthos. Illustrated in Figure 2 are the chlordane and PCB sediment concentration profiles vs. distance offshore. Resultant maximum sediment concentration at the midshelf site was approximately half an order of magnitude larger than that at the offshelf site. Exposure levels for the 100-year period were approximately five times higher than for the corresponding 10-year period for each site.

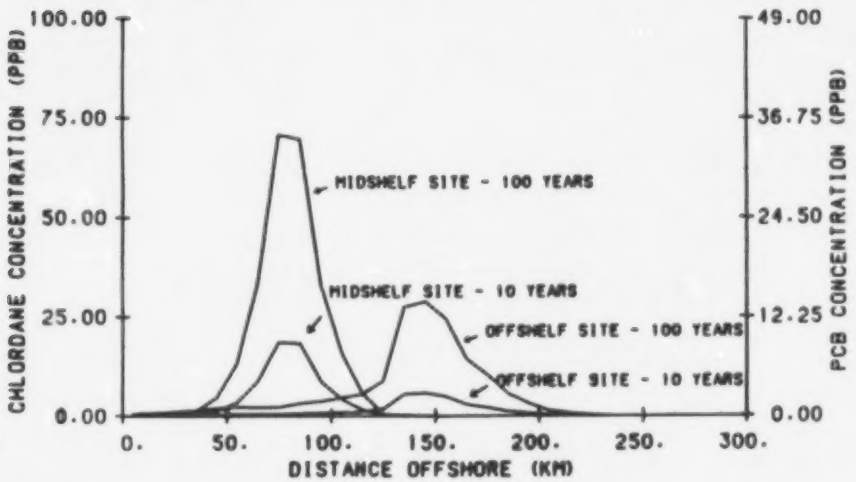

Figure 2. Chlordane and PCB sediment concentrations vs. distance offshore.

Hazard Assessment

To relate sediment concentrations to tissue residues in benthic organisms, an equilibrium partitioning model (Lake et al., 1987) currently being tested at EPA-ERLN was used, referred to as preference factor (PF) analysis. An underlying assumption in this model is that equilibrium exists between lipids in the organism and organic carbon in the sediment for organic contaminants. For neutral hydrophobic organic compounds the following equation expresses this relationship:

$$PF \quad \frac{C^i/g \text{ organism (dry wt)}}{g \text{ lipid/g organism (dry wt)}} = \frac{C^i/g \text{ sediment (dry wt)}}{g \text{ OC/g sediment (dry wt)}}$$

where PF = preference factor, superscript i refers to a specific contaminant, lipid content is expressed as percentage of organism dry weight, OC = total organic carbon in sediment expressed as percent, and C^i = concentration in respective components.

PF describes the relative affinity of a compound for organic carbon vs. lipid pool, and theoretically has a value of approximately 0.5 (Lake et al., 1987). Based on the work of Lake et al. (1987), PF values for chlorinated compounds determined in both field and lab exposures agree well with this theoretical value and provide evidence to justify this approach for certain chlorinated organics, organisms, and exposures. This model is not applicable to

contaminants that are metabolized, such as PAHs.

Because of their relative abundance along the continental shelf, slope and rise, polychaetes were used in example calculations. The percentage of lipids in polychaetes ranges from 3-4%, depending on spawning cycle (N. Rubinstein, personal communication). A preference factor of 0.5 was used for both PCBs and chlordane. Sediment concentrations were converted to dry weight by assuming a sediment solids density of 2.7 g/cm^3 and porosity of 80% (O'Connor et al., 1983). For this analysis, 0.5% organic carbon was used for the offshelf site and 1% organic carbon for the midshelf site (NYC-DEP, 1983a, 1983b).

Resultant body burdens in polychaetes vs. distance offshore for PCBs and chlordane are depicted in Figure 3.

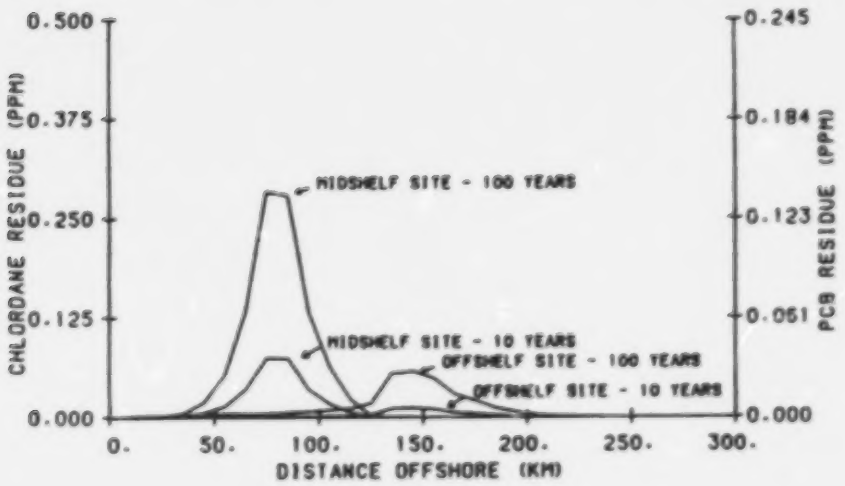

Figure 3. Chlordane and PCB residues vs. distance offshore.

After 100 years, maximum chlordane concentration of 0.28 ppm was predicted in polychaetes from midshelf disposal, compared to 0.055 ppm from offshore disposal. These values were approximately half an order of magnitude larger than the amount that the organism would accumulate in only 10 years.

Risk Characterization: Human Health

To evaluate potential human health risks based on bioaccumulation of contaminants, comparisons of potential impacts associated with midshelf vs. offshelf disposal were made in two ways. The first involved superposition of seafood landings data with projected sediment contaminant fluxes to determine potential areal extent of impacts. Then, projected body burdens for each site were compared with human health endpoints to assess the impacts from sewage sludge disposal. Although recent studies suggest that biomagnification may be an important process in accumulation of hydrophobic organic compounds in higher trophic levels (Connolly, 1987), this process was not included.

Profiles of onshore-offshore gradient of demersal seafood landings were prepared by Abt Associates Inc. for Mid-Atlantic continental shelf, slope and rise based on commercial catch data maintained by the National Marine Fisheries Service (NMFS) (Abt Assoc. Inc., 1986). Illustrated in Figure 4 is the onshore-offshore gradient in demersal seafood landings overlaid by average

concentration of sewage sludge solids during a 10-year period. As depicted in Figure 4, particulate flux from midshelf disposal would impact a higher density of demersal seafood landings than that from offshelf disposal. Thus, it is inferred that there is qualitatively higher human health risk associated with disposal of sewage sludge at the midshelf site than at the offshelf site.

To determine human health risks associated with the projected level of organism contamination as illustrated in Figure 3, the following assumptions were made: (1) polychaetes are the primary food source for commercially fished demersal species, (2) fishes from this area eat

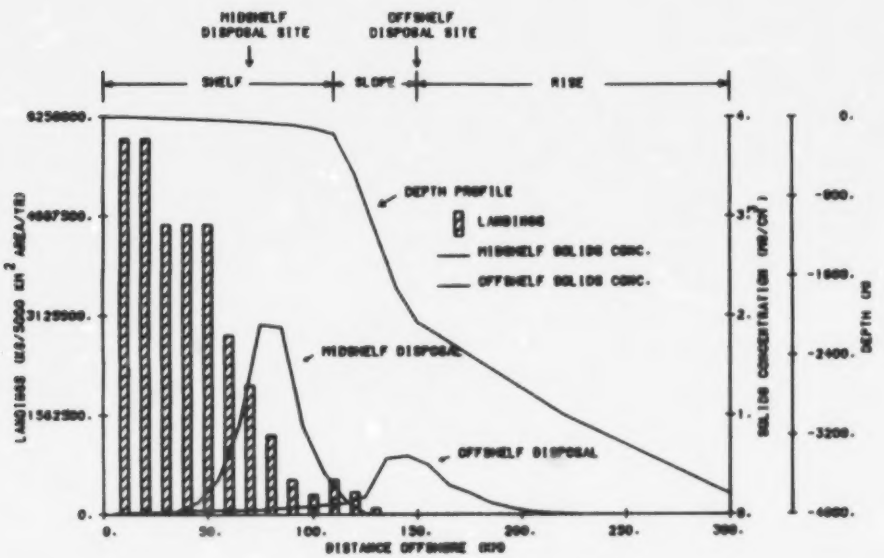

Figure 4. Demersal seafood landings data overlaid with cross-shelf profiles of average solids concentration in sediment.

contaminated polychaetes in the cross-shelf zone of maximum exposure, and (3) fishes are assumed to equilibrate in their tissues the same concentration of PCBs and chlordane that are in the polychaetes. Three human health criteria were used in this analysis: (1) the FDA tolerance level for PCBs and action level for chlordane, (2) the EPA reference dose (RfD) for chlordane and (3) EPA estimated potency factors for PCBs and chlordane.

The FDA set a tolerance level for PCB residues in edible portions of fishes and shellfish as 2.0 ppm on wet weight basis (FDA, 1984). Based on our estimates for 10 years of disposal, tissue residue levels in fish caught 80 km from shore (midshelf site) would be 0.036 ppm (wet wt), compared to a level of 0.005 ppm in fish caught 135 km offshore (offshelf site). These values increase by a factor of 5 over 100 years of disposal. Although increased levels of PCBs are associated with fishes in contact with sediment contaminated by midshelf disposal of sewage sludge, these values are still at least an order of magnitude lower than the FDA tolerance level, even after 100 years of disposal.

Based on marketability for human consumption, the FDA action level for chlordane was set at 0.3 ppm wet weight for fishes (FDA, 1979). Projected tissue residue levels from sewage sludge disposal after 10 years at both sites were at least half an order of magnitude lower than this

FDA action level, although midshelf disposal resulted in a higher value than offshelf site (0.074 ppm vs. 0.012 ppm). However, after 100 years of disposal, tissue residue levels associated with midshelf disposal approach the FDA action level; namely 0.28 ppm (midshelf). Based on these results, much higher risks to human health are associated with midshelf disposal of chlordane relative to the FDA action level.

EPA quantifies toxicological effects by assessing both carcinogenic and noncarginogenic effects. For noncarcinogenic effects, EPA uses RfD to determine whether an individual exposed to particular compound is at risk. Rfd is defined as an estimate, with uncertainty spanning approximately an order of magnitude, of the daily exposure to human population that is likely to be without an appreciable risk of deleterious effects during a lifetime (EPA, 1987a). Operationally it is derived from a no-observed-adverse-effects level (NOAEL) and an uncertainty factor which accounts for extrapolation from animals to humans. The RfD for chlordane was obtained using EPA's Integrated Risk Information System (IRIS) (EPA, 1987a). This system contains both toxicity and regulatory information on various chemicals.For chlordane the RfD is 5×10^{-5} mg/kg/day(EPA, 1985). This value is then compared to a daily dose to humans to determine whether an individual is at risk.

Daily dose to humans is calculated by multiplying projected maximum body burdens in fishes by the ratio of fish consumption to human body weight. It is estimated that an individual living in the Middle Atlantic region consumes on average 16.2 g/day of fish (EPA, 1980). Assuming that the seafood diet of a 70 kg individual consists entirely of fish from the zone of maximum exposure after 10 years of disposal, the dose of chlordane to this individual would be below the Rfd level at both sites; specifically, 1.7×10^{-5} mg/kg/day (midshelf) and 2.5×10^{-6} mg/kg/day (offshelf). However, if disposal were to continue for an additional 90 years, peak concentrations at the midshelf site, 6.5×10^{-5} mg/kg/day, would exceed the RfD; whereas offshelf disposal would still be less than midshelf disposal after 10 years, 1.27×10^{-5} mg/kg/day. Predicted exceedance at the midshelf site would occur after 70 years of disposal at the loading rate of 5.26 kg/day of chlordane. Based on these results, there is a significantly increased risk associated with disposing sewage sludge containing chlordane at midshelf as compared to the offshelf site.

For potential carcinogens, carcinogenic risk estimates are calculated using a cancer potency or unit risk value together with the assumption of lifetime consumption of fish for this particular assessment. The unit risk in fish tissue is defined as the increased cancer risk to a person who throughout life eats fish contaminated with 1 g/g of a toxicant. It is calculated by

multiplying the ratio of daily consumption to human weight by the potency factor for the contaminant. The potency factor for chlordane is 1.3 per mg/kg/day(EPA,1987b) and for PCB's, it is 7.7 per mg/kg/day(EPA,1987c). Based on these values, unit risk in fish tissue for chlordane is computed as 3.01×10^{-4} per ppm and for PCBs, 1.78×10^{-3} per ppm. To determine the increased cancer risk level, unit risk is multiplied by a contaminant residue level. Midshelf disposal of chlordane after 100 years resulted in an increased cancer risk level of 8.0×10^{-5} as compared to offshelf level of 1.6×10^{-5}. For PCBs, midshelf disposal after 100 years resulted in an increased cancer risk level of 2.5×10^{-4} versus 5.0×10^{-5} for the offshelf site. Although midshelf disposal of chlordane was associated with higher human health risks compared to FDA action level and RfD for noncarcinogenic effects, there was a higher carcinogenic risk associated with midshelf disposal of PCBs after 100 year period.

Discussion

In our application, calculations led to upper-bound estimates of risks associated with bioaccumulation of specified contaminants via the benthic food chain. Because of large uncertainties in modeling the various processes affecting transport, fate, and contaminant uptake via the benthic food chain, the results only are intended to provide estimates of the relative risk related to midshelf vs. offshelf disposal options. To predict

absolute human health risks, all sources of contamination, such as contaminant uptake via drinking water, air and other food sources must be taken into account. Since this has not been done, the risk projections in this analysis should only be compared on a relative basis and may provide a low estimate of absolute risk if used alone.

Although a formal analysis of error propagation has not been conducted, uncertainties that enter at various points in the calculation were identified. This includes uncertainties associated with human health effects, fish consumption and catch statistics data and in such processes as biomagnification and resuspension. Future efforts should address these sources of uncertainties.

Summary

A comparative risk assessment examined potential impacts associated with marine disposal of sewage sludge at midshelf and offshelf sites. Upper-bound estimates of particulate flux to bottom sediments were determined from a layered two-dimensional model. Results indicate that cross-shelf concentrations of PCBs and chlordane in the sediments were half an order of magnitude larger due to midshelf disposal compared to offshelf. Predicted chlordane residues due to midshelf disposal after 100 year period approached the FDA action level, whereas for PCBs, maximum residue levels even after 100 year disposal period were at least an order of magnitude lower than the FDA tolerance level. For noncarcinogenic risks, there was

predicted exceedance of the reference dose for chlordane after 70 years of midshelf disposal. For carcinogenic risks, the highest increased lifetime risk level was associated with PCBs and midshelf disposal. Based on these results, midshelf disposal is predicted to have higher human health risks relative to offshelf disposal.

Methods presented in this analysis have utility in a comparative risk assessment framework. However, these methods are still in their developmental phase and must undergo further testing and validation before they can become quantitative tools to predict risks associated with marine disposal of wastes. To test and validate these tools, a comprehensive biomonitoring program on regional scale should be implemented to determine actual contaminant concentrations that are accumulated in tissues of both demersal fish species and benthic organisms.

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PREFERENCES FOR NEAR-SHORE VERSUS DEEP-SEA DISPOSAL OF WASTES

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A linear programming model developed to assign multi-source wastes to a system of ocean disposal sites has been used in decision-making experiments designed to evaluate preferences for near-shore vs. deep-sea waste disposal. The model assigns wastes to sites in a way which minimizes costs of disposal subject to capacity constraints and assimilative capacity-based limits on pollutant accumulation in the marine environment. Many individuals show considerable aversion to the risks posed by in-shore pollution to human health and environmental resources, and these experiments test the extent to which such attitudes translate into preferences for deep-sea disposal when disposal costs are also taken into account.

Interactive approaches to eliciting information on preferences for the model permit users to explore through it the costs and pollution levels associated with alternative allocations of wastes among available ocean disposal sites. Experimental evidence suggests that some relatively

new, interactive optimization techniques are effective in educating users about the tradeoffs inherent in choices among technical alternatives to waste disposal problems. Current research is exploring whether such methods also have the ability to increase the acceptability of assimilative capacity-based ocean disposal management strategies, or of mixed-medium strategies which also take upland disposal options into consideration.

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RISK ASSESSMENT FOR SUBSEABED DISPOSAL OF HIGH LEVEL RADIOACTIVE WASTE

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ABSTRACT

Feasibility studies for subseabed high-level waste (HLW) disposal required development of new evaluation standards, performance assessment procedures, and models. The steps for the Subseabed Disposal Project (SDP) risk assessment are iterative as the design develops and the models and data base become more complete. First, the reference system (including transportation, emplacement, the operating repository, and possible abnormal events) is defined. Since no official radiological protection criteria exist for subseabed HLW disposal, evaluation standards are defined. Analytic and compartmental models are developed for sensitivity and repetitive risk analyses. These fast and convenient computer models are verified with numeric programs. Analysis begins with parametric studies to define the sensitivity of each input parameter. This is followed by an attenuation factor analysis to define the effectiveness of each component, and the robustness and limitations of the system. Peak individual, time integrated world population, and biota doses are computed using reference parameters. The risk is bracketed using least and most favorable input data and a stochastic uncertainty analysis is conducted when enough data are available to define probability distributions. Probabilistic accident and abnormal event risks are added to the base case risk to obtain total risk of the system. Risk assessments have been interactive with other SDP activities and were used in functional analyses for equipment and facility design, guidance of research, and feasibility evaluation. Risk sensitivity showed where the largest gains could be made in transportation and repository design and which parts of the data base should be expanded.

Two of the three iterations planned for the SDP feasibility study have been completed and examples of the results of the second iteration are given. A limited comparison is made with mined geological repositories. The limitations, uncertainties, and tasks remaining to evaluate feasibility are discussed. The summary includes the status of the risk assessment and the conclusions that can be drawn at this time.

INTRODUCTION

The US Subseabed Disposal Program (SDP) is part of the international Seabed Working Group (SWG) of the Organization for Economic Cooperation and Development/Nuclear Energy Agency (OECD/NEA), which has been investigating the technical feasibility of subseabed high-level waste (HLW) disposal since 1977. The SDP is also investigating some aspects of subseabed disposal not covered in the SWG program. The major questions in the feasibility assessment are:

1. Can a site acceptable for HLW disposal be found and characterized?

2. Can waste canisters be reliably and economically transported and emplaced in deep ocean geologic formations?

3. Are radiological risks from emplaced waste, accidents, and abnormal events below the limits for similar waste disposal processes?

The risk assessment was conducted using the seven iterative steps described in this paper.

Reference Design

The preliminary assessment completed in 1983 (NEA, 1984 and Seabed Programs Division, 1983) pointed out some weaknesses in design, and indicated where model improvements and additional data were needed. The resulting reference system used in the interim assessment completed in 1987 is shown in Figure 1. The free fall penetrator emplacement

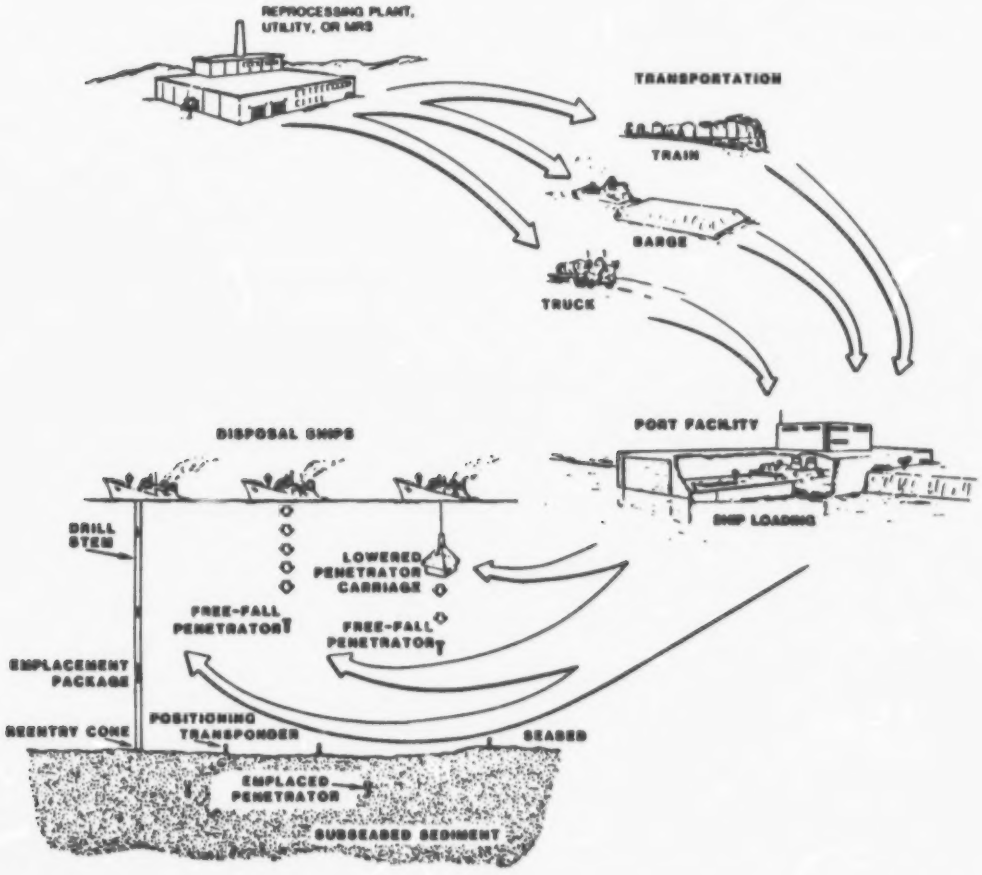

Figure 1. Reference Subseabed HLW Disposal System.

option was used in the risk assessment. Each penetrator is 6.7 m long, weighs 8200 kg, and contains 3.76 MTHM of 50 year old waste. Penetration depth would be 50 $\pm$ 20 m in all the study locations. A conceptual design of the emplacement ship was completed by the Glosten Associates (Hutchison, 1987). The ship is 160 m long, with a beam of 32 m and a draft of 8 m. The ship is designed to withstand collisions, rammings, groundings, fire, and adverse weather conditions. The structural protection system consists of a 72-mm-thick, high-yield steel hull over 6.4 m of 128 kg/m^3 urethane foam on the side and 3.5 m of 32 kg/m^3 urethane foam on the bottom. This is backed up by double longitudinal cargo hold bulkheads. Both are 38-mm-thick, high-yield steel. The collision energy absorption capacity is 3.16 x 10^6 kN-m, peak lateral force limit is 673,190 kN, membrane force is 494,330 kN, and peak lateral acceleration is 16.25 m/sec^2. This prohibits penetration of the cargo hold by vessels with speeds less than 24 knots, regardless of mass or bow construction. Protection from sinking is provided by the urethane foam. Total submerged buoyancy with all compartments flooded is 33,344 MT. The loaded ship weight is 29,595 MT, for a 12.7% buoyancy margin. The distribution of the foam is such that there is no location where cutting the ship in two would result in the sinking of a portion of the ship containing HLW cargo. The only credible mechanism for losing penetrators following a collision is through a hole made in one or two cargo holds.

Insulation from a three day 982°C fire is provided by 22.9 cm of 1430°C Fiberfrax ceramic fiber insulation and the urethane foam.

The cargo of 450 penetrators is further protected by air-cooled, shielded containers, each holding six penetrators.

Although a site has not been selected, the reference location for this risk assessment is in the Southern Nares Abyssal Plain. The capacity of the repository is 10^5 MTHM. All nuclides in HLW are included in the analyses, even though they ae not in the borosilicate waste form. Most consequence analyses were based on 1 MTHM to facilitate combining probabilistic accident and normal operation risks.

Radiological Standards

No official radiological protection criteria exist for subseabed HLW disposal. The SDP examined and evaluated the options (Kaplan, 1985a; Klett, 1986a) and developed the following evaluation standard, which encompasses or exceeds national and international regulations covering similar types of disposal. Risk is defined as the probability of an event happening times the dose from the event times 10^{-2} health effects (HE) per Sievert.

Standards for the maximally exposed individual limit are fixed irrespective of the quantity of waste. This limits the amount of waste for a given repository efficiency. Individual limits for all man-made radioactivity are:

Peak dose rate limit = 10^{-3} Sv/a

Annual risk limit = 10^{-5} HE/a.

Ten percent of man-made radioactive risk from the oceans is allocated to HLW disposal.

Standards for time-integrated world population limits allow the risk to be proportional to the quantity of power generated, so a safety factor can be assigned for each repository design:

World population dose rate limit = 10^{-4} Sv/a-MTHM

Annual risk limit = 10^{-6} HE/a-MTHM

The integration time used for the SDP is 10^5 years, which includes the peak dose rate and was shown by sensitivity studies to be the most severe. The resulting dose limit is 10 Person-Sv/MTHM. Pathways to humans include food chains, and ocean and atmospheric transport. Exposure mechanisms are inhalation, external, and consumption of desalinated water, sea salt, and aquatic foods. Usage rates for the maximally exposed individual is that of a Japanese

fisherman and the usage rates for the world population are the predicted maximum harvest rates from the oceans without depleting the species.

Models

The systems models used for the SDP sensitivity and risk analyses were either exact analytical solutions with physical and chemical limitations or compartmental models. These models have short run times, are easily modified, and are easy to input and interpret. They have been verified using numeric models for all the subseabed scenarios. TMAX and ARRAYF (Klett, 1980) were used for heat transfer analyses. The TRION series (Klett, 1987b) was used for canister life, waste leach rate, and nuclide transport through the sediment by diffusion and advection. The compartmental MARINRAD program (Koplik, 1984) computes physical and biologic transport of nuclides in the oceans, concentrations in the water, uptake by biota, and exposure and resulting doses to humans.

Sensitivity Studies

Parametric studies were conducted to define the response of the system and components to each input parameter. Sensitivity shows where the largest gains can be made in transportation and repository design, and which parts of

the data base should be expanded. It is also an indication of confidence in predictions, with higher confidence in a system with low sensitivity to variations in the input. Subseabed disposal is insensitive to most parameters. The major exceptions are the design parameter burial depth and the natural parameter vertical pore water velocity. Figure 2 shows the fraction of allowable time-integrated, world population dose vs. burial depth. With 10^5 year integration, the sensitivity is approximately -1 at 40 m. The sensitivity is much greater for depths less than 10 m. This figure shows that increasing burial depth should be a penetrator design goal and that penetration of at least 20 m is required for high confidence in the predictions. Figure 2 also shows that using too short an integration time in standards can be misleading. The steep

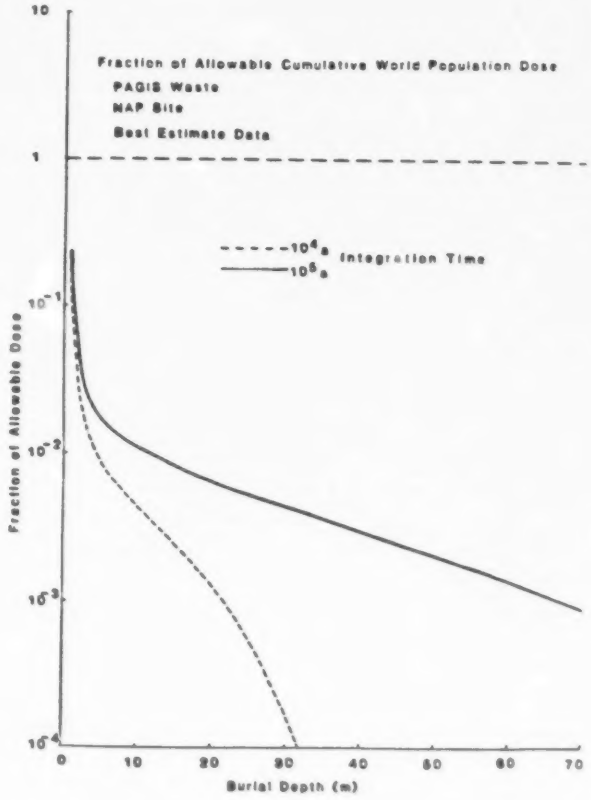

Figure 2. Burial Depth Sensitivity.

slope in the 10^4 year integration curves is caused by cutting off the analysis before the nuclides leave the sediment and is not a true indication in the reduction of total dose.

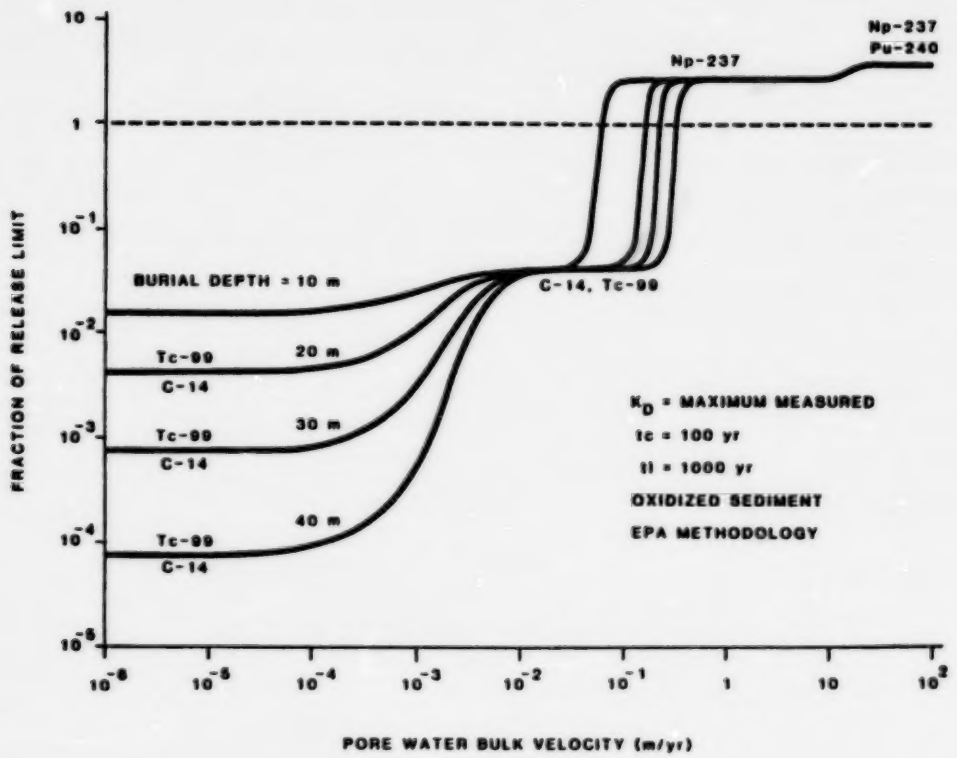

Figure 3. Pore Water Velocity Sensitivity.

Figure 3 shows the fraction of allowable population dose vs. vertical pore water velocity. It shows that if we can measure velocities as low as 0.1 mm/a and can find a site with no measurable velocity, we can conclude that there is no pore water movement or that it will have no effect on release from the sediments. The sensitivity is so large in the 10^{-4} to 10^{-2} m/a range that any change in conditions or

errors in measurement would be critical. The plateau above 10^{-2} m/a has too small a safety factor. Therefore, vertical pore water velocity less that 10^{-4} m/a should be a goal for site selection. The dominant nuclides are listed in the figure for each velocity range. The remainder of the SDP sensitivity studies can be found in Kaplan (1985b) and Klett (1987c).

Attenuation Factor Analyses

The Attenuation Factor (AF) is a measure of the reduction of flux, dose, or risk by each component and the entire disposal system (Klett, 1986b and 1987d). All components from the source to the recipient act as pathways and attenuate the risk potential. This analysis is applied to all possible source locations, recipients, and component failures. The processes for reducing risk in a waste disposal system are:

1. Retardation--Increasing the time required for waste to move between two locations.

2. Pulse Spreading--Distribution of a relatively high-intensity, short-duration release over a longer period of time.

3. Dilution--Reducing concentration by mixing waste with a large quantity of another substance.

4. Dispersion--Separating a concentrated source and distributing it in different directions.

Table I is an example including the major components in a subseabed repository.

TABLE I

ATTENUATION FACTORS FOR HLW SUBSEABED DISPOSAL
BASED ON MAXIMUM INDIVIDUAL DOSE STANDARDS
REQUIRED SYSTEM ATTENUATION FOR 10 REPOSITORIES = 8.5×10^{18}
(NAP, 5000a CANISTER, 2500a LEACH, 50m BURIAL DEPTH)

	Scenarios			
Component	Complete System	Failed Canister and Waste Form	On Sea Floor	Repository Alone (without oceans)
Canisters	2.9×10^{2}	—	2.9×10^{2}	2.9×10^{2}
Waste Form	2.6×10^{1}	—	2.6×10^{1}	2.6×10^{1}
Sediment	1.6×10^{5}	1.2×10^{9}	—	1.6×10^{5}
Oceans	1.3×10^{11}	1.3×10^{11}	5.5×10^{12}	—
Total	1.6×10^{20}	1.6×10^{20}	4.3×10^{16}	1.3×10^{9}

The Complete System column shows the relative importance of each component and that the system has a large safety factor. The next column shows the robustness of the system. Peak dose rates do not increase even if the canister and waste form fail completely. The short-lived nuclides that

would normally decay in the canister decay in the sediment and the pulse spreading of the leaching waste form is negligable compared to the spreading by diffusion through the sediment. On Sea Floor shows the AF of canisters dumped on the sea floor. The safety factor is too small, even with only 10 repositories. These AFs are also an input to accident analyses. The last column shows that the oceans must be included in SDP risk analyses because the attenuation of the repository alone is not sufficient. This is typical of all repositories. Although the AF method of performance assessment was developed for nuclear waste disposal, it can also be applied to toxic waste disposal.

Risk Analysis

Figure 4 illustrates the types of computations that are made in an SDP risk analysis. Each analysis computes

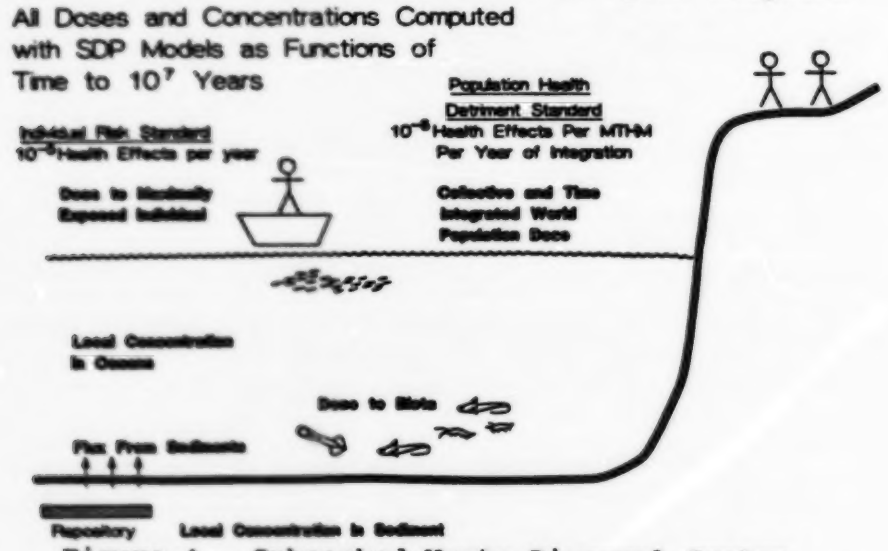

Figure 4. Subseabed Waste Disposal System.

concentrations in the sediment, oceans, and aquatic food; flux from the sediment; doses to biota, the maximally exposed individual, and the world population. The types of analyses include:

1. Reference design emplaced waste.

2. Probabilistic transportation and emplacement accidents.

3. Bracketing of 1 and 2 used the most and least favorable input data.

4. Stochastic uncertainty when there are enough data to define the distributions.

5. Consequence analyses of abnormal scenarios.

6. Total probabilistic risk for the system.

Figure 5 shows the total maximum individual dose and the dose from each nuclide as a function of time for emplaced waste using best estimate input data. Only a few nuclides with low distribution coefficients significantly contribute to peak dose rates. It also shows that the dose must be integrated at least 10^5 years to obtain meaningful results. Similar curves are generated for each of the above output parameters and types of analysis.

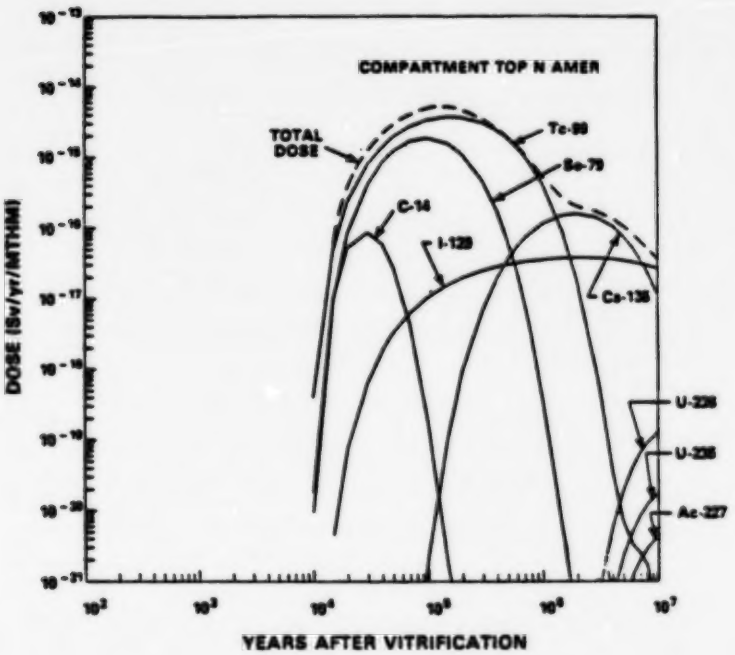

Figure 5. Individual Dose vs. Time in the Top N. American Compartment, by Nuclide -- Post-Emplacement Release at 5,000 Years, Mean-Risk Parmaeter Values

The results of peak individual dose studies from emplaced waste are summarized in Table II.

TABLE II

PEAK INDIVIDUAL DOSE -- POST-EMPLACEMENT RELEASE
(UNDAMAGED CANISTERS, 50-YEAR-OLD WASTE)

RISK LEVEL	TIME OF CANISTER FAILURE (Yr)	PEAK INDIVIDUAL DOSE $\left(\frac{Sv/yr}{MTHM}\right)$	TIME OF PEAK DOSE (Yr)	COMPARTMENT	PRINCIPAL NUCLIDES AND % CONTRIBUTION	PRINCIPAL PATHWAYS AND % CONTRIBUTION
Low	10^4	1.0×10^{-18}	7.0×10^6	Top N. Amer.	I-129(91%) Cs-135(9%)	Seaweed(94%) Fish(4%)
Mean	5×10^3	5.2×10^{-15}	1.5×10^5	Top N. Amer.	Tc-99(71%) Se-79(29%)	Mollusc(29%) Crustacean(28%) Seaweed(24%) Fish(19%)
High	3×10^2	7.3×10^{-13}	7.0×10^4	Top N. Amer.	Se-126(55%) Tc-99(40%)	Fish(31%) Seaweed (30%) Crustacean(23%) Mollusc(16%)

The compartment refers to the location in the model responsible for the peak dose rate.

Sea transportation risk with the new emplacement ship is negligible compared to the risk from emplaced waste. Mean values for the probabilistic peak individual dose per MTHM from shipping is 4.4×10^{-20} compared to 5.3×10^{-15} Sv/a for emplaced waste. A corresponding comparison for collective population dose is 4.4×10^{-12} to 2.2×10^{-2} Person-Sv.

Of the abnormal scenarios investigated, damaged emplaced canisters, enhanced leach rate, and changes in ocean currents had almost no effect. If all distribution and position coefficients were 0, doses would increase by a factor of about 300. If pore water velocity was 1 m/a and the canisters failed, the doses would increase by a factor of 5×10^{7}. Since probabilities could not be assigned to abnormal events, they were not included in the total probabilistic risk.

Total probabilistic risk safety factors for a single 10^{5} MTHM repository for both types of standard and three levels of input data are given in Table III.

TABLE III

Data Level	Peak Individual	Collective Population
Most Favorable	1.0×10^{9}	1.2×10^{8}
Mean	1.9×10^{5}	4.6×10^{2}
Least Favorable	1.4×10^{3}	1.9

In this table, an allowable individual risk of 10^{-6} (10% of total man-made) from nuclear waste disposal was used. Collective population safety factors are not dependent on quantity.

The data base and complete 1987 risk assessment are in de Marsily (1987) and Klett (1987a).

Interpretation and Application

Sometimes it is helpful to compare disposal risks to parameters other than criteria or regulations. Figure 6 shows how low the individual dose from a subseabed HLW repository is compared to background doses for each of the data levels. Comparison to other methods of disposal is also useful. A literature search (Narga, 1985) showed that predicted peak individual doses for mined geologic repositories of 10^{5} MTHM in salt, shale, clay, and granite ranged from 1.2×10^{-10} to 1.1×10^{-3} Sv/a, which averages higher than the 5.2×10^{-10} Sv/a for subseabed. Predicted

collective population doses integrated 10^5 years from a mined geologic repository in tuff (US EPA, 1986) ranged from 10^2 to 10^3 higher than subseabed, depending on waste leaching assumptions.

The results of the 1983 risk assessment were a great help in planning the research program, selecting site criteria, and performing the functional analyses for design of the penetrator and emplacement ship. As an example, the 1983 risk assessment revealed several weaknesses in the reference ship design that resulted in transportation accidents contributing the highest risk to the system.

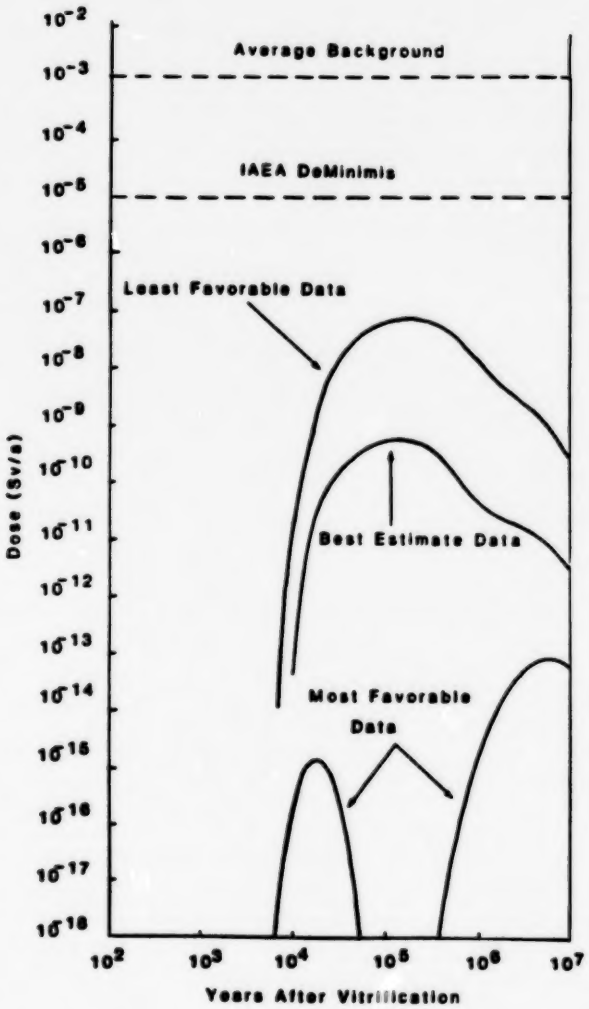

Figure 6. Doses to Maximally Exposed Individual from a 10^5 MTHM Repository.

This information was used in the functional analysis and functional requirements studies for the current ship design. Peak individual doses from shipping accidents were reduced by a factor of 1.8×10^6.

The technical feasibility study is not complete and there are still some uncertainties in the SDP risk assessment. These are:

- Data base is not complete.
- All transport-related processes may not be known.
- Biological and ocean modeling are not complete.
- Probabilities of abnormal events are not known.
- Probabilistic risk analyses are based on estimated distributions because of data limitations.
- A method of retrieving emplaced canisters has not been developed.
- Sites are not completely characterized.
- Models are verified but not validated with field tests.

There are also some site limitations, which are:

- A site with greater than 0.1 mm/a vertical pore water movement is unacceptable.

- A site with dislocations may be unacceptable.

The status of the SDP risk assessment after the second of the three planned iterations in the feasibility study can be summarized as follows:

- Basic transport models from emplaced waste to humans are completed. Accurate predictions can be made with analytic and relatively simple numeric models.

- Sensitivities of all known input variables are defined.

- Interim risk assessment predicts good safety margins with conservative standards.

- Risk- and sensitivity-based functional analyses have led to penetrator and emplacement ship conceptual designs that meet cost and safety requirements.

- Sites have been found in the Atlantic and Pacific that passed initial screening tests.

- Transport properties of all sediments tested indicate that sediment is an effective barrier.

- A waste package can be built that will survive long enough for recovery following an accident and during the thermally active period following disposal.

- Major physical and biological pathways to humans have been identified.

All analyses to date indicate that subseabed would be a safe method of HLW disposal and that predictions could be made with a high degree of confidence.

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JUSTIFYING SEA DISPOSAL OF WASTES

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Introduction

It is hard to explain why the controlled disposal of wastes at sea, by dumping or incineration, has caused so much controversy. Over the past 3 years, the activities of environmental organizations in publicly denouncing these practices, have been far more aggressive than they were prior to international agreements designed to prevent pollution by waste disposal at sea. With little regard for balance or perspective, and without firm scientific evidence to support such action, this persistant and well-orchestrated lobby has already influenced a moratorium on low-level radioactive waste disposal and is currently working to prevent the use of incineration at sea. To the outside observor, it might well appear that the role of international bodies in determining the application of sea disposal technologies has been usurped by independent non-governmental organizations.

Yet the record clearly shows that since the establishment of the London Dumping Convention and its regional (North-East Atlantic) equivalent, the Oslo Convention, dumping and incineration have contributed to a relatively minor extent to marine contamination or pollution. Significant localized impacts have been identified in certain poorly-dispersive environments but, wherever possible, measures have been taken in such cases to reduce inputs or to find more suitable dumpsites. In general the range and quantity of industrial wastes dumped has been falling, while efforts are being made to improve monitoring and to tighten the justification process.

Under the circumstances, it is discouraging that outside interests seem now to be threatening to undermine the work of the Convention in defining procedures and criteria by which the oceans can play their role in waste management without damage to marine resources. For more than a decade, many

scientists and administrators have committed themseleves to the concept that an active, multi-disciplinary and international body offers the best opportunity for a harmonized approach to protection of the marine environment against pollution from anthropogenic activities. For me it is ironic that these individuals should now be cast in the role of transgressor. Nevertheless, reality dictates that if sea disposal of wastes is to have a future it will be necessary to provide a degree of justification which is seldom, if ever, required of other waste disposal practices. The idea that the oceans deserve preferential status in terms of protection has strong support.

I believe it is very appropriate that an international meeting of scientists concerned with the technology of waste disposal at sea should consider these issues at this time. Scientific principles will play a large part in the justification process but scientists must find more opportunities and more effective mechanisms for communicating their views to the regulators and to the public. In this paper I try to present my perspective on some of the most important elements of the sea disposal debate.

The London Dumping Convention

I think it is quite fair to say that the London Dumping Convention, and the various regional agreements on the prevention of marine pollution, have achieved a considerable measure of success by reducing indiscriminate use of the seas for waste disposal. Much of this progress was achieved in the early seventies, even before many countries had ratified their membership. The introduction of a harmonized licensing system, the classification of wastes according to potential hazard and the designation of dumpsites, were probably sufficient to ensure that the worst effects of uncontrolled dumping could be averted. So the question that many have been asking is "Where does a dumping convention go after its initial implementation?"

According to the articles of the London Dumping Convention, which is now ratified by 61 countries, there is plenty of work to be done. In addition to maintaining an inventory of dumping statistics contributed by its members, and resolving difficulties which arise with regard to interpretation and implementation of the Convention, members are expected to assist in the establishment and implementation of regional protocols on dumping, to transfer information and technology between themselves and to regional bodies, and to generally promote protection of the marine environment against pollution from all sources. And these activities have, indeed, provided the Consultative Meeting, its advisory body the Scientific Group on Dumping and the Secretariat with sufficient material to keep them well occupied over the last 12 years. It has to be appreciated that, as origionally drafted, the text of the London Dumping Convention did not attempt to provide a comprehensive guide to the control of dumping at sea. There were a number of key terms which needed to be defined, in particular the terms "trace quantities" and "rapidly rendered harmless" which are associated with exemptions to the black list (Annex I). It was also necessary to draft regulations and a code of practice governing the incineration of wastes on board vessels at sea, and to study legal aspects of certain articles which relate to jurisdiction, liability and disputes. In recent years, the Scientific Group on Dumping has drafted detailed guidelines on the application of Annex III of the Convention (which lists the factors to be considered in issuing permits), on the criteria which should be used in assigning substances to the black or grey list (Annex I or II) and, recently, on the control of dredged material disposal. These guidelines contain practical advise to assist the regulatory agencies of member countries in meeting the requirements of the Convention. Some would criticize the fact that guidelines are not mandatory, but for many countries the incorporation of new requirements into national law is a time-consuming exercise and, since guidelines are subject to continual up-dating, it would not

be practical to do this. Nevertheless, countries are expected to implement guidelines adopted by the Consultative Meeting as if they were a part of the Convention.

This synopsis of work within the London Dumping Convention does, I believe, show that progress has been made since its foundation. And I do not accept the harsh or even derogatory comments which are sometimes made concerning the lack of progress within the Convention. Bearing in mind that multi-national and multi-disciplinary groups of scientists and administrators, all of whom work only part-time on matters related to dumping at sea, meet two or three times per year at most, the performance has been respectable and significant. Nevertheless, some observors have expressed the view that the Convention appears to be losing momentum and that it lacks cohesion of purpose. I have some sympathy with the latter view and feel this is an opportune moment to diagnose the ailment.

Defining objectives and measuring progress

Although the London Dumping Convention continues to function and develop, there is no generally accepted yardstick by which its success can be measured in the long term. The reason for this is that there is no unanimity amongst Convention members on the real objectives of the Convention. On the matter of the general exhortation contained in the pre-amble and articles, that members should act together to prevent pollution of the seas from all sources, and in particular from dumping, there can be no disagreement. But the fundamental question is whether this should be achieved by progressively working towards the elimination of dumping, or by ensuring that dumping does not damage the marine environment or marine resources. To the technical reader, there can be very little doubt that the text of the London Dumping Convention, and indeed the regional Oslo Convention and related protocols, inherently recognizes that there are

circumstances under which wastes may be disposed of at sea without harmful effects. Implicit in this interpretation is that where damage does not occur, pollution has not taken place. I am quite certain that almost everyone involved with the management of wastes would wish that this interpretation was universally acknowledged. However, the fact that it is not means that the development within the Convention of a self-assessment procedure is not only difficult, it is a veritable minefield.

If success were to be gauged by the absence of harmful effects from dumping or incinerating wastes at sea, the critical data base would be that which stems from monitoring dumpsites and regional sea areas. On the other hand, if progressive reduction of the number and variety of wastes dumped or incinerated were to be the criterion, then success could be judged solely on the basis of annual dumping statistics. In practice, information is compiled from both sources and member countries are divided on the relative importance of the two assessment procedures. Part of the difficulty is that returns on dumping and monitoring are not complete and thus neither their individual nor their relative merits can be assessed accurately. Clearly this is an outstanding administrative problem which needs to be resolved as a matter of urgency since no agreement on the principle of self-assessment is likely to emerge if the supporting evidence is lacking.

The alternative proposition to that which equates pollution with environmental damage is that pollution results from any input of wastes to the environment. Much has already been written in the scientific literature concerning the definition of pollution and the distinction between pollution and contamination. I will not add to this debate except to say that, as a scientist whose work involves trying to bridge the gap between the protection of the environment and the needs of society, I have great difficulty with the suggestion that wastes cannot be accommodated within the ecosystem

without harmful effect. I share the view of the founding fathers of the Convention, or what I believe to be their view, that the practical definition of pollution contained in Article I recognizes that it is possible to dispose of waste and other matter in the sea in a manner that is not "liable to create hazards to human health, to harm living resources and marine life, to damage amenities or to interfer with other legitimate uses of the sea." The opposite interpretation would render most of the other articles superfluous, a situation which I find too ludicrous to contemplate.

Having nailed my colours to the mast, I must immediately stress that I do not regard the distinction between contamination and pollution as clear-cut, nor do I suggest that scientific proof that harm has not occurred is always possible. But I do maintain that environmental managers, with appropriate scientific advice, can and should decide, on the basis of the criteria laid down by the Convention, what is an acceptable disposal operation and what is not. Indeed, I firmly believe that this was, and is, the basis for international agreement and international action in the area of marine pollution control. However, there should be no impediment whatever to the implementation of measures at the national level, by appropriate national regulatory agencies, which transcend the requirements of the Convention and which reflect a preference for land-based solutions to waste disposal. Articles IV (3) and VII (5) of the London Dumping Convention clearly acknowledge this right.

The reality of the situation at present is that, for some states, ideological and political pressures make it difficult to accept dumping or incineration at sea as legitimate waste disposal technologies. The Nordic states, in particular, have pledged themselves publicly to seeking an end to these practices world-wide. Predictably this has caused sharp polarization amongst contracting parties to the London

Dumping Convention and, just as seriously, it has largely undermined the relevance of scientific advice in the decision-making process. In my estimation, this situation will not be resolved at statutory meetings of the convention under the present climate - positions are too entrenched for this to happen. Unfortunately, I have to say that, if the usefulness and integrity of the convention are to survive in the long term, I see no alternative to formal renegociation of the agreement involving, inter alia, clarification of its objectives. While this may seem a rather drastic move, perhaps it would be a good thing if international agreements were periodically evaluated, amended and up-dated in the light of experience; an opportunity, if you like, for rejuvenation.

Waste management strategies

In the meantime, countries that wish to retain the right to dump wastes at sea will find themselves under increasing pressure to justify their activities and decisions in this regard. Rather than resist such pressures, I feel that the opportunity should be taken to emphasize that sea disposal should not be viewed in isolation, but that it should be seen as only one of several options assessed under the terms of an integrated waste management strategy. The principles underlying a waste management strategy may be broadly segregated into the philosophical and the scientific, and no two jurisdictions are likely to give precisely the same weight to one category or the other. However, in the international arena it is the technical and scientific principles which offer the most hope for agreement and, for this reason, the Scientific Group of the London Dumping Convention has begun to examine certain concepts and principles relevant to waste management with a view to their possible incorporation into the annexes of the Convention and associated guidelines.

On a number of occasions the Consultative Meeting of the London Dumping Convention has recognized the importance of adopting measures which will control marine pollutants at source. This clearly implies support for the widely acknowledged and eminently sensible hierarchy of waste management which gives preference to low-waste and no-waste technologies and urges greater use of treatment prior to disposal of wastes. There would seem to be no reason why the hierarchy could not be embodied more formally in the Convention, for example by including it in Part C of Annex III.

Concepts and principles in waste management

An important principle under review by the Scientific Group is that which states that, having made a comparative assessment of all disposal options available for a waste, the one selected should be the least detrimental to Man and the environment. Two points, in particular, should be made in relation to this principle. The first is that its use depends on a thorough examination of all feasible options, and so its adoption is contingent on all options remaining open - in other words a decision to prohibit an established disposal technology would render the principle unworkable. The second point is related to the first and this is that, in the context of the London Dumping Convention, situations might arise where the disposal option of least detriment would not be permissible under the Convention. Such situations are already envisaged for contaminated sediments dredged from certain harbours. The Consultative Meeting has asked the Scientific Group to study this matter further but, clearly, resolution of the difficulty will require some surgery to the articles and annexes of the Convention.

But perhaps the most important concept in the development and implementation of controls on waste disposal is that of "assimilative capacity". What is meant by this is that each sector of the environment has some finite capacity to absorb

or accommodate waste without causing significant or irreversible changes within the indigenous biological community, or damage to the resources it contains. Acceptance of the assimilative capacity concept is implicit in Article 1 of the London Dumping Convention and explicit in the pre-amble where reference is made to the limited extent of the sea's capacity to assimilate wastes. From the technical point of view, the existence of assimilative capacities for individual substances can be demonstrated using a variety of criteria to determine impact - however, the acceptability of an impact is very much a matter for society. For some substances the assimilative capacity may be so small that it would have very little practical value. Despite these limitations, the concept provides a basis for selective and judicious use of the marine environment for waste disposal within an holistic waste management strategy. In its 1986 report, the Advisory Committee on Marine Pollution (ACMP) of the International Council for Exploration of the Sea (ICES) provided a detailed discussion on assimilative capacity and, in the same year, GESAMP (Group of Experts on the Scientific Aspects of Marine Pollution, Working Group No. 23) provided practical advice on the definition of assimilative capacity in a variety of situations.

Recognizing that there is still considerable controversy over the use of assimilative capacity as a management tool, the ACMP has suggested that it could find greater acceptance when combined with a set of principles elaborated by the International Commission on Radiological Protection (ICRP). The principles, known as Justification, Compliance and Optimization can be applied to the regulation of any of Man's activities which potentially impact upon the environment or human health. They are also compatible with the hierarchy of waste management and should be incorporated into waste management strategies.

In brief, justification requires that no practice should be adopted unless there are clear net benefits to society; compliance requires the establishment and observation of exposure limits for products, raw materials and wastes; and optimization requires that exposures should be kept as low as reasonably achievable, taking technical, social and economic factors into account. ACMP considers optimization to be of special relevance to the use of assimilative capacity and to the identification of the least detrimental disposal option. Firstly, optimization, by taking into account social factors including public perception of harm, will help in defining an acceptable level of change or effect; secondly, optimization will help to establish the disposal route and the disposal method which causes minimum disturbance. To quote from the ACMP report : 'Not to couple a comprehensively based system of optimization to the application of the assimilative capacity concept would seem to be little short of folly. Only when optimization is incorporated into the use of the assimilative capacity concept can it be ensured that the least detrimental option is identified'.

The importance of monitoring

In my view, taken together, the hierarchy of waste management, the ICRP principles, assimilative capacity and 'the option of least detriment' provide a practical as well as a conceptual framework within which dumping at sea can be justified. But is it sufficient to justify a practice simply because it fits into a grand scheme for environmental management? From the scientific viewpoint the scheme may be acceptable, but the public has grown wary of scientists and administrators and will tend to resist approaches which go against what are perceived to be the best interests of society. But if it is true that the majority of environmental scientists believe that an holistic approach to waste management is necessary for the sake of the environment and human health, and that to eliminate the sea

as a disposal medium could cause more serious impacts than would otherwise be the case, then scientists must play a leading role in support of their position. Quite simply, the scheme must be seen to work.

Scientists and engineers have a role to play at all levels and at all stages of environmental management. They provide the data and information on which decisions are made, standards are set and performance is monitored. Without adequate research no system for managing chemical substances can be justified. In this context, I would emphasize the importance of monitoring the marine environment as part of research programmes rather than as routine activities. Whether the objective is to study localized effects of individaul dumping operations, or the far-field effects of multiple inputs, monitoring will always be most productive if it is tailored to the location, innovative, and cognizant of experience. At the same time monitoring must be cost-effective or it will mitigate against sea disposal, and it must be guided by the effect-parameters established during the optimization process used to define the assimilative capacity. Because monitoring is an essential element in justifying disposal of wastes at sea, the results of monitoring should be compiled on a regular basis, presented as far as possible in a non-technical format and made available through national and international channels. If national administrations which permit sea disposal operations do not take this approach it will be increasingly difficulty to justify continuation of this practice.

The global perspective

The sea, like the atmosphere, is a resource shared between nations. In using the sea for whatever purpose, each country should ensure that it does not infringe the maritime rights of another. This is the underlying rationale for the London Dumping Convention and related international agreements. Indeed, it is the recognition that acceptable changes to the

marine environment due to waste inputs cannot be established unilaterally, that has fostered the development of harmonized approaches to dumping. But although the sea is a global resource, it is also an integral part of the human environment of coastal states each of which has its own combination of needs and circumstances. Each country must decide on the precise approach to waste management which it will adopt, taking into account many factors including the requirements of international agreements to which it is party. In this way a balance is reached between the needs of the state and the wider, multi-national interest. Thus, in addition to its vital role in a comprehensive scheme for waste management, the disposal of wastes at sea is justified by the very existence of an international forum to debate the issues and to define mutually-acceptable criteria for protection of the marine environment world-wide. I believe the preservation of this forum should be a matter of universal concern. Without it the cause of a fully international approach to waste management will face a set-back from which it may be difficult to recover.

Finally, and briefly, I return to where I started out. The point at issue is not whether or not dumping and incineration at sea can be carried out without causing harm to the marine environment, it is whether or not environmental organizations have adopted the correct approach to eleviating public concern over the mismanagement of chemicals in our environment. The concern is that national administrations have failed to control the products, processes and wastes of industry in the best interests of society and the environment. Funnily enough, people are probably more tolerant of what happens in their backyards than what happens in the sea. They see the backyard, understand it and have some control over what happens in it. For the most part they neither see, understand nor have any control over what takes place in the sea. From Jacques Cousteau to Greenpeace, the

western world has been bombarded with images of the sea as a delicate, vulnerable and intolerant environment - an ultimate source of food and perhaps a last refuge for humanity.

As a weapon to attack the system, sea disposal is an easy option to select. But I would ask environmental organizations is it in fact the correct one? There is not the faintest possibility that anything approaching the quality of life which people wish to lead can be sustained without producing wastes and not all of these wastes will be usable in a productive manner. River inputs to the ocean have been, and will continue to be, significantly more influential on the quality of the marine environment than dumping or incineration are ever likely to be. Evidence is mounting that selective, controlled disposal of certain wastes in the marine environment is the preferable environmental option. The quality of the terrestrial environment is likely to change more rapidly due to anthropogenic activities than the quality of the sea. Countries which now feel that they can accommodate all wastes on land will find that, in the long term, there ability to do so is progressively reduced.

International organizations committed to protecting or managing the environment, whether governmental or non-governmental, can achieve much that is good for society and the environment both at the regional and global levels. In my view cooperation rather than conflict will produce the best results.

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